STATE OF THE UK'S WOODS AND TREES 2025

Trees and woods in a changing world

WOODLAND TRUST

Technical Report

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Contents

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State of the UK's Woods and Trees 2025: trees and woods in a changing world

Woods and trees are woven into our identities and sense of place, as individuals and communities across the four nations of the UK. In *State of the UK's Woods and Trees 2025*, you'll find out how woods and trees need our help to build ecological resilience to thrive for wildlife and people.

The climate is changing with devastating consequences. Woods and trees help us to adapt and reduce the impact of a changing climate by providing a wide range of social benefits and environmental services.

Despite tree cover rising to 13.5%, woodland biodiversity continues to decline. For instance, the woodland bird index was 37% lower in 2022 than in 1970 and has decreased by 15% in the last five years. This is largely due to our woodlands not being in good enough ecological condition.

Trees and woods also improve our health and wellbeing. However, these benefits are not equally distributed. Following on from our groundbreaking tree equity mapping, new research also indicates that lower income areas not only have less tree cover, but existing woods in these areas are not as effective at boosting local people's wellbeing. This is because they contain fewer biodiversity traits shown to have a positive impact.

This report sought to find out why we continue to ignore the deep interdependencies between trees, wildlife, climate and people and why we cannot afford to keep doing so.

By collating and analysing a huge range different sources of data and evidence, we found:

- The UK continues to miss woodland creation targets – The Climate Change Committee's Seventh Carbon Budget published earlier this year, calls for woodland creation to be nearly doubled by 2030. Yet just 45% of targets have been met in the UK in the last four years, which means we've missed out on an additional 8.5 million tonnes of carbon dioxide absorption by 2050.
- Woodlands face an escalating and interacting suite of threats, yet the scale and urgency of action is lacking – for instance, currently the UK hosts 121 introduced native tree pests. An estimated £919.9 million is spent each year in the UK on managing only six pests.
- Hanging over and exacerbating all these threats is the impact of climate change such as drought, wildfires and extreme weather events.

What can we do to turn things around? The most recent Bunce survey led by the Woodland Trust and UKCEH and published in 2024, gives an unparalleled 50-year insight into change in Britain's woodlands. This revealed a decline in ecological complexity and biodiversity due to a lack of management, and therefore less resilience.

What this means is we desperately need sensitive management of UK woodlands to improve their ecological condition and unlock their ability to adapt to climate change. Ecological restoration of woodland habitats through sensitive management not only supports nature recovery but also supports a well-functioning carbon cycle that provides a stable, long-term store of carbon. Only with increased levels of sensitive management can woods and trees continue to provide benefits for people and wildlife.

You are invited to explore the current state of UK woods and trees via this technical report. It puts you at the cutting edge of science and evidence for native woods and trees: the state they're in, the threats they face and reasons to be hopeful. Also available is an online interactive report and four reports summarising the evidence in each UK country.

We are only five years away from 2030, the year for many targets to be delivered: to halt the decline in species, to end deforestation and protect 30% of land for nature. The latest *State of the UK's Woods and Trees* report tells us there is an urgent need to:

- Enhance, expand, connect and protect native woodlands.
- Improve the evidence to better understand how we can help them to help us.
- Invest in the future through financial support and training.

This report is both a celebration of native woods and trees and a rallying cry to protect, improve and expand them across the UK. In so doing we will be restoring, healing and nurturing ourselves, as we are not separate from nature. Communities, organisations and governments must work together with individuals from the full spectrum of society to build a stronger and more resilient natural world.

Extent

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Introduction

From the Caledonian pinewoods of Scotland to south England's lowland beech woods and including globally rare temperate rainforest, trees and woodlands are varied and valued parts of the landscape of the United Kingdom. These vital habitats for a large number of species provide a range of ecosystem services essential to our landscape function, economy and human health and wellbeing (Quine et al., 2011; Sing, Ray and Watts, 2015, other chapters this report).

UK woods have faced different policies and land use practices throughout their history and have been significantly shaped by these. Following widespread clearing during the First and Second World Wars, British woodlands were mainly planted and managed for timber, leading to a focus on non-native conifer plantations which still dominate much of the landscape today (Mason, 2007).

Lowland, native broadleaf woodlands have also faced significant change over time, becoming increasingly fragmented, and many small, privately owned broadleaf woods have fallen into poor condition due to a lack of management and loss of traditional techniques such as coppicing (Hopkins and Kirby, 2007).

In addition, climate change, development and land use change, tree pests and diseases, invasive species, pollution and deer damage all continue to impact our woods today (Fuller and Gill, 2001, Reid et al., 2021, later chapters this report).

The combination of these historical and current impacts has led to a dramatic shift in the cover, composition and distribution of the UK's woodlands.

To ensure woodlands can withstand and adapt to the threats and challenges they face, we need to continue to protect and restore those we currently have, as well as creating new woods. Increasing the cover of our woodlands adds to resilience by providing greater buffering, more transition zones and corridors and larger areas where species can thrive. Expanding woodland cover requires a comprehensive understanding of where they currently are; woodland cover statistics allow us to see and track this over time.

The UK has set woodland cover targets: to be planting 30,000 hectares a year by 2025 and increase woodland cover to 16.5% by 2050. Both are currently failing to be met (see creation chapter this report).

In State of the UK's Woods and Trees 2021, the woodland extent figures we reported at that time showed woodland covered 13.2% (3.2 million ha) of the UK's land surface. This was split roughly equally between native and non-native species, with ancient woodland making up 2.5% of the UK's land area. Here we provide an update on these figures using the National Forest Inventory (NFI) and Northern Ireland Woodland Register. It is important to note that the figures for total extent do not provide insight into the quality of the woodlands, or their ability to provide services to people and the environment to help resilience and adapt our landscape to the changing climate. These other aspects of woodlands are explored further in this report.

About the data

Woodland is defined in the UK forestry statistics as l*and under stands of trees with a minimum area of 0.5 hectares and a canopy cover of at least 20% or having the potential to achieve this*. This definition relates to land use, rather than land cover. This means that open spaces and felled areas that are awaiting restocking are included as woodland.

Datasets are constructed using a combination of earth observation satellite imagery, data from planting grants, aerial imagery and information from organisations that manage woodlands (including statutory bodies). They are updated annually, with 2024 being the most recently published editions.

The data does not specifically state whether woods are native or non-native, but record whether tree canopy species are broadleaf or coniferous, which is a useful surrogate as our native tree species are typically broadleaf, while many of the non-native species grown for commercial forestry are coniferous. An exception to this is in some areas of Scotland where the native conifer Scots pine (*Pinus sylvestris*) is present.

Canopy cover in the UK also includes trees outside woods (TOWs) as well as larger parcels of woodlands. TOWs include copses, hedgerows, trees on farms, wood pastures and parklands. Trees outside woods contribute significantly to the total canopy cover and often provide essential corridors between woodland patches. Data on TOWs has only been updated for England since 2021.

Ancient and veteran trees, temperate rainforest and urban trees are discussed in separate sections of this report.

Estimates of woodland cover

The total area of woodland in the UK in 2024 is estimated to be 3.28 million hectares or 13.5% of the total land area (Table 1). Of this total, c. 1.3 million hectares (41%) is in England, c. 0.1 million hectares (4%) is in Northern Ireland, c. 1.5 million hectares (46%) is in Scotland, and c. 0.3 million hectares (9%) is in Wales.

Table 1: Woodland area by UK country in 1998, 2020 and 2022 (million hectares) and in brackets as a percentage of total land area.

Year	England	Northern Ireland	Scotland	Wales	UK Total
1998*	1.24 (9.5%)	0.08 (5.8%)	1.30 (16.6%)	0.30 (14.5%)	2.92 (12%)
2020	1.31 (10%)	0.121 (8.7%)	1.47 (18.8%)	0.31 (14.9%)	3.21 (13.2%)
2024	1.34 (10.3%)	0.12 (8.6%)	1.50 (19.4%)	0.31 (15%)	3.28 (13.5%)

*1998 is selected as the baseline year because figures for England, Wales and Scotland have been revised to produce estimates that are consistent with subsequent data from the National Forest Inventory, which are therefore comparable to 2020 and 2024 figures.

Of this, conifers account for around 48% of the total woodland area, although this proportion varies from 71% in Scotland to 22% in England (Table 2).

Table 2: Area of woodland (million hectares) by forest type 2024. Source: Forest Service, National Forest Inventory.

Forest type	England	Wales	Scotland	Northern Ireland	UK
Conifers	305	139	1070	64	1578
Broadleaves	1033	173	441	55	1701
Total	1338	312	1511	118	3279

Ancient woodland extent

Ancient woods, by definition, have developed over centuries and are known to be one of our most diverse terrestrial habitats. The definition varies across the UK: in England, Wales and Northern Ireland, it refers to land that has had continuous woodland cover since at least 1600, while in Scotland, the threshold is 1750. This is because planting of trees was uncommon at this time and reliable maps are available from these dates, giving more confidence to these being natural, native woodlands. This long history means that ancient woodlands have a rich biodiversity and are referred to as being irreplaceable.

The extent and distribution of ancient woodlands are recorded on the ancient woodland inventories of each country. These serve as crucial records of where these remnants of functioning landscapes still exist. However, interpretations vary significantly between countries regarding what qualifies as ancient woodland.

The inventories are owned by the statutory nature conservation body in each country (Natural England, NatureScot, Natural Resources Wales) with the exception of the Northern Ireland inventory, available from the Woodland Trust. In Scotland, additional categories of woodland are included: 'long-established woodland of plantation origin' (LEPO) and 'other Roy' woods (present day woodlands which appeared on the Roy maps but not on the OS first edition). Table 1 shows the current Ancient Woodland Inventories data.

Since SoWT 2021 there has not been a complete update of any of the inventories and so currently the data is the same (Table 3).

Table 3: Estimated area (ha) of ancient woodland across UK countries and % of total land area Source: Ancient Woodland Inventories.

Woodland type	England	NI	Scotland	Wales	UK
Ancient woodland	364,200 (2.8%)	2,700 (0.2%)	148,150 (1.9%)	94,940 (4.6%)	609,990 (2.5%)
LEPO and 'other Roy' in Scotland		7,270 (0.5%)	204,610 (2.6%)		211,880 (0.9%)
Total	364,200 (2.8%)	9,970 (0.7%)	352,760 (4.5%)	94,940 (4.6%)	821,870 (3.4%)

Updates to the Ancient Woodland Inventory

The existing ancient woodland inventories are essential for planners and developers, policy makers, landowners, foresters, conservationists, landscape historians and many others keen to protect and restore these special wooded habitats. However, due to a lack of updates since the original inventories, many important fragments of ancient woodlands may be missed. Updates are therefore vital. Despite strong planning protection policies, ancient woodlands are being lost or damaged by development or inappropriate management simply because they are not recorded on the inventory.

England

The AWI in England was originally produced in the 1980s, without the benefits of computerised mapping techniques. The original AWI inevitably contains many omissions and inaccuracies, and perhaps more crucially, it did not include any woods smaller than two hectares or ancient wood pasture and parkland sites. There have also been significant steps taken to restore some ancient woodlands damaged by conifer plantations, yet these positive changes are not always recorded. The basic methods for identifying ancient woodland have not changed, but the policy, public awareness, technology, expertise and knowledge has increased significantly, making a full update both more feasible and more urgent.

In 2019 the Ancient Woodland Inventory update project began. This project, led by Natural England and with significant financial support from the Government and the Woodland Trust, is identifying all ancient woodlands greater than 0.25ha in England. For the first time it also includes ancient wood pasture and parkland sites. The new AWI uses high resolution Light Detection and Ranging (Lidar) maps to reveal features of ancient woodland, as well as geo-referenced old maps and ecological site surveys to identify new sites, county by county in England. The work is being completed using Local Environmental Record Centres (LERCs), meaning their vast expertise and local knowledge contributes to the quality of the final product.

The updated AWI for the completed counties can be viewed and downloaded at Ancient Woodland - Revised (England) - Completed Counties | Natural England. For all other counties which are yet to be updated please see Ancient Woodland (England) | Natural England Open Data Geoportal (arcgis.com). At the time of writing, the updated counties have added over 7,000ha of ancient woodland to the inventory. The inventory is due to be completed in 2026.

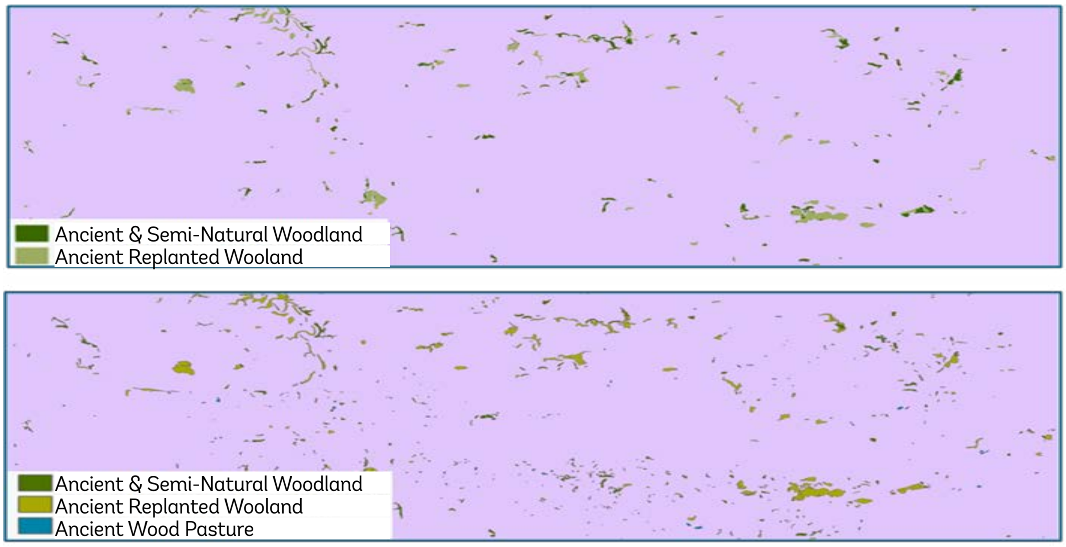

Figure 1: Snapshot of the Original AWI (top) and the updated AWI (bottom) from Central Devon. Natural England 2023.

Northern Ireland and Ireland

Ancient Woodlands Ireland is a research project funded by the Department of Agriculture, Food and the Marine (DAFM) and the National Parks and Wildlife Service (NPWS). This four-year (2024–28) all-island project seeks to update the inventories of ancient and long-established woodlands in Ireland and Northern Ireland and is a collaboration between Maynooth University, Teagasc, Dundalk Institute of Technology and the Woodland Trust Northern Ireland.

Despite their importance, ancient woodlands are now a rare and fragmented feature on the island of Ireland, and major gaps in our understanding of the extent and condition of ancient woodlands in Ireland exist.

Ancient Woodlands Ireland is using a multidisciplinary approach to address these knowledge gaps. Researchers in Teagasc are using novel machine learning approaches to digitise woodland cover from historic OS maps from the 1830s. Historians on the team are delving deeper into manuscript maps and written records of land cover from the 16th to the 19th century. Paleoecologists are examining preserved pollen in soil cores to reconstruct vegetation history and determine woodland age at sites of conservation importance. Ecologists and microbiologists will investigate whether certain plants and soil microorganisms can be used as indicators of ancient woodland in Ireland.

Ultimately, the Ancient Woodlands Ireland project will provide valuable data that can be used to enhance the protection of Ireland's ancient woodlands and help prioritise sites for ancient woodland restoration.

Wales

The AWI in Wales was updated in 2021 with changes to the previous version from 2011 which includes additional evidence presented to Natural Resources Wales (NRW) through public enquiries. This encompasses private estate maps, tithe maps, management records and photographs. The evidence was reviewed by a panel of NRW experts and the inventory amended if the evidence indicated that a site has had another land use for a significant period, i.e. woodland is absent from the site in two consecutive epochs of OS historic maps, or there has been significant ground disturbance to the site, e.g. quarrying, mining or development. Amendments have included removal of sites from the inventory, adjustments to boundaries or changes to the ancient woodland category of sites. The 2011 version remained static until it was replaced by the 2021 version. The 2021 version is being updated at least annually as new evidence becomes available.

Scotland

In Scotland the provisional dataset was published in 1987 and there has been no update since then. The Scottish government has made a commitment to produce an accurate record of where ancient woodland is, and NatureScot is convening a steering group to begin this process. Woodland Trust Scotland will be an active member of the steering group and has already collated feedback from across its team to identify ways to improve the tool.

It is critical to protect these precious habitats now, and understanding where they are is the first step. Improved mapping helps achieve this.

Trees outside woods

In the 2021 *State of the UK's Woods and Trees* report, data was reported that showed there was an estimated 742,000ha of tree cover outside woodland in Britain (no data exists for Northern Ireland). This is 19.4% of Britain's total canopy cover and 3.2% of total land area, increasing the total canopy cover in Great Britain to 3,719,000ha. Since around 94% of TOWs are native broadleaved species, this represents as much as 30% of the total native tree cover in Britain.

In early 2025, a map of England trees outside woodlands (single trees, groups of trees, and small woodlands) was published, showing that they make up 30% of England's tree cover. The map allows for more targeted tree planting and woodland creation efforts by showing where these smaller areas of trees could be connected to nearby wooded areas.

Conclusions

Canopy cover is a fundamental measure of the state of the UK's woods and trees, against which we can track progress with achieving goals for woodland expansion. While extent figures cannot tell us anything about woodland condition, understanding their location and size can help in improving connectivity and size to increase wider landscape resilience.

Recent data shows that woodland cover across the UK is increasing. However, these increases are only very small, and the UK's current cover of 13.5%, is well below the EU average of 38%. We need to expand and connect up our current woodlands through increased woodland creation.

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Woodland ecological condition

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Introduction

In Great Britain (no data exists for Northern Ireland) our woods are monitored for their ecological condition to help inform management decisions. Having woodlands in good condition is vital as they provide habitats for wildlife, access for people's wellbeing, and ecological functioning. Woodlands in good condition are more likely to be able to provide these services, be resilient and able to adapt to future changes in climate (Bellamy et al., 2018).

Ecological condition is measured as individual attributes (listed below) that can be important in informing management decisions. These attributes can also be aggregated to give an overall condition score that links to ecological complexity. Ecological complexity has been defined as 'the number of components in a system and the number of connections among them.' Components can include, for example, species, height classes, functional groups and habitats. Connections can include species interactions, energy flows among species, or connectivity among habitat patches' (Bullock et al., 2022). Greater complexity can lead to greater resilience and ability to adapt due to this variety of components contributing to ecosystem functioning, providing greater buffering and stability (Bullock et al., 2022).

In the first *State of the UK's Woods and Trees* report, we reported the Forest Research National Forest Inventory's woodland ecological condition data showing that just 7% of Britain's native woodlands are currently in good ecological condition (Reid et al., 2021). However, the majority of woodlands fall into an intermediate condition category. Understanding what this means for practical management and improving condition requires looking at the individual attributes that make up condition. Using this data and new data from the Bunce survey (Smart et al., 2024), we take a closer look at the findings and their implications for practical management and policy.

Methods

Here we combine information from Forest Research's National Forest Inventory (NFI) and the Bunce project to gain insight into the condition of our woodlands.

Forest Research's NFI is designed to provide information about the size, distribution, composition and condition of our woodlands and how these change through time. This is vital for developing and monitoring the policies and guidance that support the sustainable management of woodland. Woodland ecological condition (WEC) is ascertained by conducting a statistical assessment of 15 indicators of woodland ecological condition and a further classification into its condition status of favourable, intermediate and unfavourable condition.

The Bunce survey monitored woodland change over 50 years and opportunistically provides data for eight of the 15 NFI condition attributes (note, they are not collected in the same way or on the same timelines).

These 15 indicators (highlighted in bold below) can broadly be split into three categories around structure, composition and threats.

Structure

Regeneration within and around stands: abundant regeneration is vital for securing the future of woodlands as maintenance of woodlands requires turnover of species. We know the UK's native tree species have a wide genetic diversity, and naturally regenerated trees show adaptation to local conditions that may help them survive better than planted trees. Natural regeneration can also create a more natural species mix, which can be beneficial for wildlife.

Deadwood volume (m3 per ha): deadwood and veteran trees are important parts of a varied structure in a woodland. Deadwood is perhaps better referred to as decaying wood as it is anything but dead - an estimated 500 arthropod individuals live in each litre of wood from dead branches, and an average of 2,500 arthropods inhabit each kg of 'wood mould'. Decaying wood can be present as dead attached parts, dead standing trunks, hollowing trunks, or on the woodland floor. Decaying wood plays a disproportionate role in maintaining ecological processes and supporting species as well as providing essential nutrients that can be used again by trees for growth. Given the high ecological value of deadwood, the UK Woodland Assurance Standard (UKWAS) makes recommendations for deadwood management as part of its certification standards for sustainable woodland. Generally, the more deadwood of a variety of sizes within a woodland, the greater the biodiversity value.

Vertical structure: Having variation in vertical structure in a woodland is important. This is because impacts from threats such as disease and windthrow affect different age classes and sizes of trees. For example, Dutch elm disease tends to attack mature trees over 20 years old. Having variation in structure and an age distribution of tree species, provides more resilience to these threats within a woodland. The provision of gaps and open spaces allows for more regeneration which provides the crucial structural and age diversity needed.

Veteran trees: Veteran trees are mature trees that have developed valuable decaying wood features, not necessarily as a consequence of time, but due to their life or environment. Veteran trees may not be very old, but share similarities with ancient trees, such as trunk or branch hollowing, or significant amounts of other decaying wood. Veteran trees provide a variety of microhabitats. For example, tree cavities provide nest and roost sites for birds and bats. In Europe, an estimated 30% of forest-dwelling birds use tree cavities, and it is well known that the lack of availability of cavities of a range of different types is a limitation for bird populations.

Composition

The composition of a woodland is important for its resilience. A greater diversity of species is more likely to be able to resist or adapt in response to change (Bellamy et al., 2018). This is because a more diverse woodland provides functional equivalence/redundancy. This is where multiple species can share similar roles in ecosystem functionality, for example fixing nitrogen, pollination, decayer or scavengers. Therefore, if one species is removed or reduced due to perturbation, another can continue to fill the vital functional role. Many measurements of condition are looking for species diversity within woodlands. This includes field and ground flora and the number of native tree and/or shrub species.

In addition, the size of the woodland is important. A larger size allows for more species diversity and greater buffering of the habitat. The proportion of habitat being woodland also allows for greater diversity as there is more habitat for woodland species.

Nativeness of occupancy is also important for the composition as native species have co-evolved with other native species that leads to greater diversity. For example, in the UK we know that 2,300 species (birds, fungi, insects and mammals etc) are associated with sessile oak and English oak (*Quercus petraea* and *Q. robur*), of which 326 are obligate and it would require a greater number of other tree species to support them (Mitchell et al., 2019).

Threats

Some of the attributes measured are threats rather than a measure of current condition attribute. This is important as it allows us to spot and respond to threats before they become too severe and impact condition and resilience long term. Reducing these threats should be a primary objective, as other management interventions to help improve condition cannot be effective if the site is still facing major threats e.g. creating gaps for regeneration is futile if deer numbers are so high that any regeneration gets eaten. In condition assessments **herbivore impact, tree health** and **invasive species** are all measured. In the UK there are high numbers of native deer and some non-native deer that eat regenerating seedlings and saplings. At this very high density, deer can all but dramatically reduce any tree and shrub growth (see deer section of this report). The UK also hosts a large number of pests and diseases that can kill off high numbers of trees (see tree pests and pathogens section of this report). Our woodlands are also hosts to invasive species such as grey squirrels which damage trees (see grey squirrel section of this report) and rhododendron which shades out and outcompetes other woodland species (see ancient woodland restoration section of this report). Measuring these threats allows for early and appropriate interventions.

While the results in this section are based on surveys (NFI and Bunce) that provide a picture of condition across Great Britain, the same set of indicators can be used to assess the condition of an individual wood to inform management decisions (e.g. FC/Sylva Woodland Condition App).

	NFI	Bunce
Open space	Most native woodlands fall into the unfavourable category for open space, with around 50% of native stands in Britain having less than 10% open space.	Open spaces (glades and paths) became less common up to 2021, with the biggest reduction between 1971 and 2001.
Regeneration	Most woodlands fall into the 'intermediate' category for regeneration with only 9% of stands recording no presence of seedlings, saplings or young trees – however to count as having regeneration only requires the presence of just one of either seedlings, saplings or other young trees.	The number of quadrats where regeneration was recorded declined over the 50 years for ash, birch, sycamore and hazel. Ash is the second most common species found regenerating in woods though, apart from holly which is higher now.
Age distribution of tree species	No woodland type exceeded 19% in favourable condition for age distribution, meaning that the majority of native woodlands have only one or two age classes present (out of young, intermediate or old). 80% of native stands in Britain have a restricted number of age classes with only two or fewer distinct age classes of trees.	From 1971 to 2022 there was a shift to fewer, larger trees across the surveyed woodlands: in 2022 there were only 40% of the number of stems of 1971 but those stems were about 2.8 times as large in basal area.
Deadwood volume	80% of native woodland habitats scored unfavourably for deadwood volume: 46% of native woodland stands have no deadwood within them, 25% have less than 10m3 per hectare, 26% of stands contain between $10m^3$ and $100m^3$ and 3% have over $100m^3$ per hectare.	From 1971 to 2022 plots with some type of deadwood present increased from 60% to 90%.
Veteran trees	98% of native woodland stands have less than 0.05 veteran trees per hectare.	NA
Vertical structure	52% of native stands are in favourable condition for vertical canopy structure. 71% of native stands have three or more distinct canopy stories.	NA
Vegetation	Most native woodlands in intermediate condition	Ground flora richness declined by around 30% from 1971 to 2001, then increased by around 10% from 2001 to 2022 resulting in an overall decline of around 20%.

	NFI	Bunce
Nativeness of occupancy	80% of native stands have less than 5% non-native species in the upper canopy.	The broad character of the woodland canopy has not changed, with 11 of the most common tree species appearing in the same rank order as they did 50 years ago. Most species were less frequently detected in 2021 than in the previous surveys, because of stand thinning and reductions in stand density as woods have aged. This is typical following disturbance and in 2021 British woodlands are now made up of fewer but bigger trees. Two shade tolerators however – beech and holly - have notably increased. Holly regeneration in plots has increased consistently over the 50-year period, particularly in sites with lower mean winter minimum temperatures and those sites where these winter temperatures have increased most. The result suggests that much of the increase in holly is explainable by a shift to warmer winters, particularly in parts of Britain that have warmer winters overall.
Number of native tree/ shrub species	84% of native stands are in favourable condition for 'nativeness' of canopy, with 68% of native stands having four or more native tree species per stand in Britain. Scots pine plantations and native pinewoods are largely unfavourable for the number of native tree species, which means they only have up to two different tree species present.	NA
Size of woodland parcel	Most native woodlands fall into favourable condition for size of woodland, with 66% of native woodland stands found in woods of < than 100 hectares in size.	NA

	NFI	Bunce
Herbivore damage	40% of native woodland habitat is in unfavourable condition for herbivore damage. Around 50% of woodland area has signs of herbivore browsing damage below 1.8m in Britain; 47% in England, 26% in Wales and 59% in Scotland.	Signs of sheep, red deer, cattle and horses were generally uncommon across years and reduced from 1971 to 2021. However, signs for 'other deer' were commonly noted and increased markedly over time, suggesting that a range of deer species constitute the greatest herbivore presence in the woodland sample. Deer signs were noted in 33% of plots by 2021.
Invasive species	9% of native woodland area has unfavourable status for invasive species. 10% of native woodland stands have invasive species present.	NA
Tree health	3% of native woodland area has unfavourable status for pests and diseases. And 4% of native woodland stands have pest and diseases present.	Ash trees were present in 44% of plots and 49% of these showed signs of disease. Elm has also declined due to Dutch elm disease.

Results

Discussion

Investigating both the NFI and Bunce data shows some worrying trends for our woodlands and their current condition. There is a reduction in structural complexity of our woodlands, likely as a result of a lack of intervention, and mitigated only slightly by storm and disease events which introduce gaps and remove trees. In addition, composition of woodlands is also becoming simplified, likely as a consequence of a lack of structural diversity because of insufficient dynamism and variability in light levels, resulting in a reduction of micro-climates and niches for plants to colonise and thrive, in turn leading to a reduction in micro-habitats and resources for woodland wildlife. The role of deer in reducing both structural and compositional diversity is high. The increase in herbivore damage correlates with increased deer numbers since the 1970s (see deer chapter of this report for more detail) and is having a major impact on the condition of our woodlands.

Conclusion

The benefits that woodlands provide for biodiversity, recreation, flood alleviation, health and wellbeing, soil health and carbon sequestration are delivered much more effectively when the woodland is in good condition. This is crucial, because healthy ecological systems provide vital services for people and support woodland wildlife, and we need to understand what practical steps can be taken to improve those woods in poor ecological condition.

Data from condition assessments shows the need for urgent action. The historic and recent events that have seen woodlands fall into poor condition urgently need to be addressed. Appropriate management is needed to move woodlands back into good condition (see management chapter in this report).

Evidence gaps

- Better measurement of appropriate management and how this relates to condition. As noted in the management chapter of this report, the current measurements of management are not very holistic and don't paint a good picture of condition.
- Further monitoring of the effectiveness of management interventions on addressing key aspects of condition is necessary to ensure actions produce desired outcome.
- More regular updates of NFI WEC. More sharing of this data will allow other organisations to undertake a more detailed assessment of condition. As the majority of woodlands fall into the intermediate category, being able to look at this in more detail would allow a better understanding.
- Adding a measurement of what direction, improving or not, to condition assessments.

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Ancient and veteran trees

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Introduction

Our oldest and most special trees have an extraordinary ability to resonate with the hearts and minds of people young and old. They act as custodians of our cultural heritage and guardians of our biodiversity. The UK's oldest and most special trees include our internationally renowned collection of ancient and other veteran trees, as well as trees termed heritage trees. While many heritage trees are ancient and veteran, not all are, and heritage trees are often characterised just as much by their cultural or historical importance than by their age or ecological value. It is therefore useful to clearly define what is meant by ancient, veteran and heritage trees. This is particularly important because several key pieces of planning policy make explicit reference to ancient and veteran trees and the value of their irreplaceability. Therefore, the language we use to describe our oldest and most special trees is crucial.

Ancient trees have developed beyond maturity into the ancient life phase. They are old in comparison to other trees of the same species. Ancient trees can be **chronologically ancient** having reached great age, and/or show signs of being **developmentally ancient** with distinct developmental changes to their morphology, such as the gradual and episodic retrenchment and reorganisation of the crown, or the fragmentation into individual functional conductive units.

Veteran trees are mature trees which, due to their life or environment, have significant decay features (a physical attribute they share with ancient trees) but are neither developmentally nor chronologically ancient.

Ancient therefore describes the life phase of an old tree, whereas veteran describes a tree of high ecological value due to its deadwood habitat. All ancient trees are veteran but not all veteran trees are ancient.

While there is no single established definition of **heritage trees**, they are the exceptional, extraordinary, uncommon and unexpected trees in our landscapes. They can be champions renowned for their great size, have outstanding cultural or historic significance, or be ancient and veterans in their own right.

PHILIP FORMBY/WTML

The Grantham Oak – a pedunculate/ English Oak (*Quercus robur*) that is more than 500 years old

Ancient tree development and architecture

Ancient and veteran trees often exhibit significantly more complicated forms than their younger counterparts. As trees age from seed to sapling, and eventually from mature to ancient, they pass through a series of developmental changes. These developmental stages are described in the **life stage model** (figure 1), which can be simplified into three stages: young, mature and ancient (although there are actually many more stages in the full model). Furthermore, the developmental process is non-linear, trees may revert in developmental stages and different sections of the same individual tree may also be in different developmental stages.

Intrinsic factors governing a tree's development, such as its individual genotype and the species evolutionary history, provide a blueprint for a tree's form, while extrinsic constraints, such as environmental variability, resource availability and competition, also interact with these intrinsic factors to govern a tree's architecture.

Stress inducing factors, e.g. drought, may also influence a tree's architecture. Reorganisation of crown architecture therefore, represents a critical stress tolerance strategy of all trees.

Throughout the long life span of ancient and veteran trees, individual trees may have gone through several cycles of progression and regression through the life stage model, losing part of their crown through subtractive growth and retrenchment and gaining new parts of their crown through new iterations and growth. Although this process is dynamic, it plays out in tree time and can be difficult to observe.

This ability to reorganise and restructure, combined with the modular nature of tree growth, the reiteration of similar repetitive architectural units (stems, leaves and branches etc), is an important part of the puzzle to understanding the longevity, resilience and adaptive capacity of our ancient and veteran trees. Reorganisation confers resistance to an array of stressors and explains the complexity and diversity of forms observed among our oldest trees. Understanding the complex and dynamic morphologies and architecture of ancient and veteran trees is critical in underpinning their active management and conservation. Identifying the morphological and structural traits that aid trees in developing into the ancient phase may better help us support this transition in future ancients.

ILLUSTRATIONS: CAROLINE MIEKINA

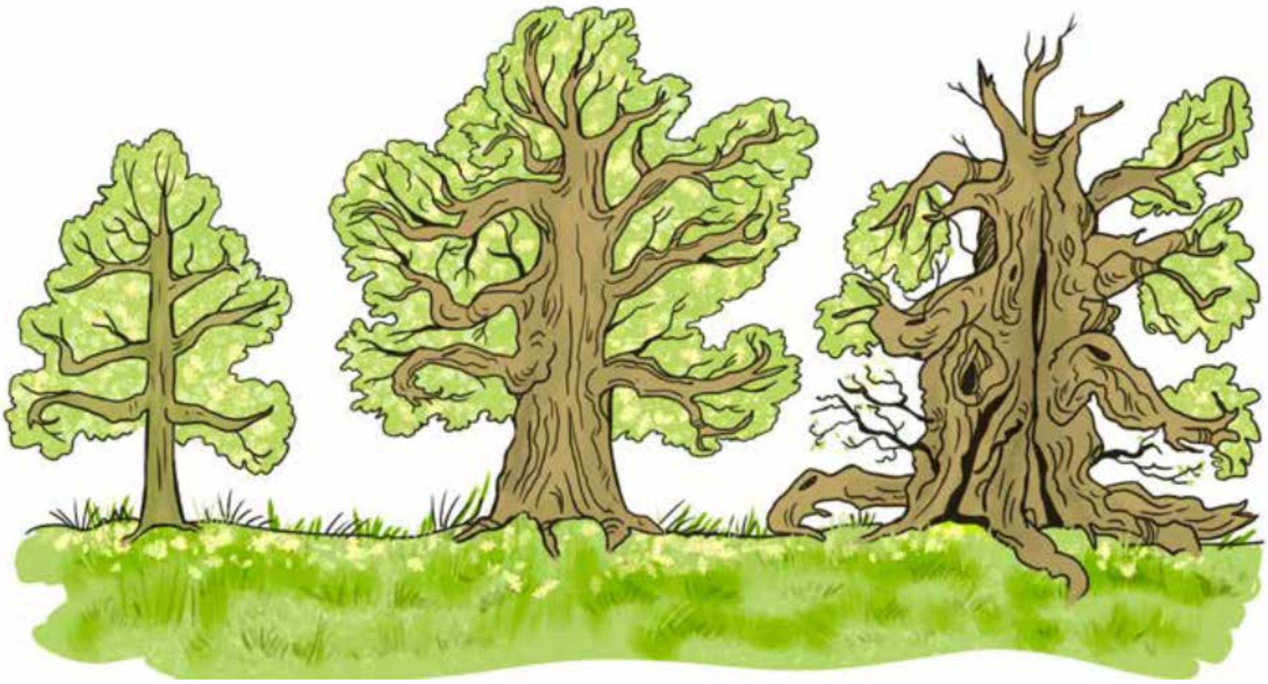


Figure 1: Simplified illustration of the life stage model: while the model is presented linearly it's important to remember development and aging in trees is a complex and dynamic process

SCATTER

The Woodland Trust funded a groundbreaking research project called **SCATTER**. SCATTER stands for SCanning Ancient Trees with TERrestrial LiDAR. The project created detailed 3D models of 40 ancient and veteran oaks from across the UK, mapping their structure, size and form in mm scale detail.

Each ancient oak was scanned using a LiDAR (light direction and ranging) instrument. The LiDAR instrument sends out hundreds of thousands of laser pulses per second and when these hit something (parts of a tree for example), the 3D location of these hits is recorded. A full tree scan is then made up of a cloud of millions of these 3D points. Using a computer algorithm these points are turned into a complete 3D model of the tree. We can then use the 3D model to calculate the volume of the tree i.e. how much wood it contains (which tells us how much carbon it stores), and other important things like the size, shape and location of individual branches, as well as the size and position of microhabitats.

The outputs of this project have allowed us to create an open access digital archive allowing anyone, regardless of ability or location, to access these natural wonders. It also provides a permanent scientific record of these trees should they be lost in the future. Alongside this we can compare ancient oak tree architecture from across a spectrum of different contexts and ages (such as management history, location and oak demographics), giving insight into what makes the architecture of our oldest trees so special. We are actively engaging with practitioners to actualise potential benefits to tree managers from implementing these technologies which could be used to develop management plans or understand better interventions.
Open-source models available at: https://sketchfab.com/SCATTER.project

Ancient and veteran trees are keystone structures in combating the climate and biodiversity crises

Our ancient and veteran trees, and their associated soils, are irreplaceable ecological **keystone structures** in our landscapes, meaning that they disproportionately support biodiversity and ecosystem function in relation to their small area of occupancy and biomass of individual trees (Manning et al. 2006). Such ecological functions include **carbon sequestration and supporting biodiversity** through the provisioning of **microhabitats.** In landscapes and ecosystems fragmented and disrupted by human activities and management, large, old and scattered ancient trees are **biological legacies**. While ancient and veteran trees are often found in woodlands and forests, they frequently exist as old, scattered trees in a wide variety of landscape and land use contexts from medieval deer parks, wood pasture, agricultural land and even developed urban spaces. Old, scattered trees are often the oldest structures within highly disturbed landscapes and provide integral ecological continuity, acting as isolated refugia reminiscent of past landscapes for myriad specialist and rare species. Through their resilience to environmental perturbation, ancient trees can provide a "life-boating" effect, supporting the persistence of other species. In addition, they can play a pivotal role in **nature recovery** and provide nucleation sites for **ecosystem recovery** after disturbance (Manning *et al.* 2006).

As keystone structures, our oldest and most special trees have a central role in combating the **climate crisis**. Our ancient trees are winners of the life history lottery; being sessile, they have stood fast against past climatic extremes and environmental disturbances and have demonstrated extraordinary resilience in the face of a changing climate. Because of this, they are a vital genetic reservoir of resilience to future adverse climates and represent an invaluable, genomic resource **buffering the genomic diversity and adaptive capacity** of tree populations to future extreme climate perturbations (Cannon *et al.*, 2022). In addition to buffering tree populations against climate change, they also provide insulating microclimates within their microhabitats that buffer the consequences of severe events such as drought, or extreme cold for dependent species and thus offer species essential climate refugia (Lindman *et al.,* 2022). They also provide ecosystem services such as transpiration, which modifies local microclimate and provides a cooling effect. Living for several centuries, or in some cases several millennia, ancient trees represent carbon sinks over significant time scales. The soils beneath our oldest trees are often overlooked, and they too represent a substantial carbon sink, with soil organic carbon accumulation being disproportionately greater under large trees (Dean *et al.*, 2020). Furthermore, carbon sequestration is linked with the age and size of trees; many of our oldest and largest trees therefore disproportionately sequester carbon compared to their younger, smaller counterparts (Mildrexler *et al.*, 2020).

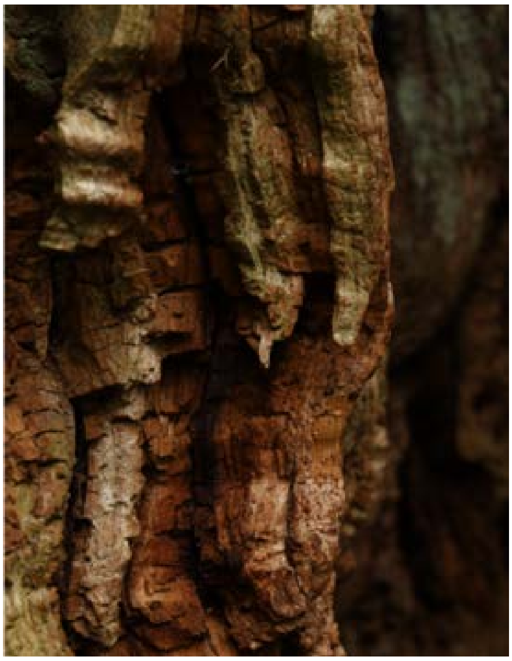
Old, microhabitat-rich trees are an invaluable resource in combating the **biodiversity crisis**. They provide habitats for a wide range of plants, animals, fungi and microbes including birds, mammals, invertebrates, mosses, lichens, bryophytes and fungi. Many of which are highly specialist and rare, only occurring within the unique and irreplaceable niches provided by ancient and veteran trees. Our ancient and veteran trees are internationally important reservoirs of threatened biodiversity.

For example, the threatened and legally protected rare oak polypore fungus (*Buglossoporus quercinus*), almost exclusively fruits on exposed heartwood of ancient open-grown oaks in wood pasture and deer parks – the open-grown old oaks of Windsor Great Park are the global stronghold for the species (Crockatt et al., 2010). Lichens are complex slow growing symbiotic organisms and are highly specialised; many threatened species of lichen thrive on the continuity provided by our oldest trees. The beech marble lichen *(Pyrenula nitida*) occurs on the smooth-barked rain tracks of beech and hornbeam, typically in old-growth pasture woodlands or wood pasture, with only two known current extant locations in the UK (New Forest and Burnham Beeches; Sanderson 2024).

Our ancient and veteran trees provide the long term spatial and temporal ecological continuity these species require by acting as colonisable stepping-stones of habitat that increase ecosystem connectivity in space and time. As many species dependent on these trees have limited ability to disperse, if there is insufficient availability of ancient and veteran tree habitats due to ancient and veteran trees being rare or isolated in the landscape, or if sites are lacking ancients of the future, it will have negative consequences for these ancient and veteran tree dependent species. Some species, such as the endangered violet click beetle (*Limoniscus violaceus*), require the continued presence of old hollow trees (Cuff et al., 2020). The rarity of suitable trees in the landscape combined with the beetle's poor dispersal ability are major contributors to its current decline. Additionally, many sites with ancient and veteran trees suffer from a problem known as the **generation gap**, where there is a large age gap between the current cohort of ancient and veteran trees and the ancients and veterans of the future (Read and Bengtsson 2021). For example, Sherwood Forest where there is an age gap of several centuries between the oldest trees and the next generation. This threatens the temporal continuity of habitat for species dependent on ancient and veteran trees.

Deadwood microhabitats and saproxylic assemblages of ancient and other veteran trees

As trees age, they accumulate damage and dysfunction, and this leads to the accumulation of decay habitats and unique, complex structural elements that are not apparent in younger trees. **Deadwood is a misnomer and it would perhaps be better referred to simply as decaying wood**. So-called deadwood provides **refugia**, **substrata** and a **nutrient source** for a wide array of organisms. Approximately 25% of all forest species are dependent on deadwood for at least part of their lifecycle and we call these species **saproxylic** (Stokland et al., 2012). Despite the importance of deadwood, changes in forest management and the decline of ancient and veteran trees have led to a scarcity of this resource in our landscapes. As a result, large diameter deadwood has undergone massive reductions in its availability since the industrial revolution (Dahlberg et al., 2010). As such, many saproxylic organisms are now at risk – for example 18% of European saproxylic beetles are considered threatened (Cálix et al., 2018). Individual ancient and veteran trees provide a disproportionate amount of these decay microhabitats, such as **cavities**, **hollowing**, **sap runs**, **bark shelters** and more, with several often co-occurring on the same tree, providing a diversity of habitats and microclimates that allows high levels of biodiversity to flourish. These decay habitats, such as heart rot (figure 2), may take exceptionally long to develop, for example hollowing in oak trees typically begins to develop at around 200-300 years, and by 400 years almost all oaks have significant hollowing and heart rot (Ranius et al., 2009). This makes the deadwood habitats provisioned by ancient and veteran trees irreplaceable. Other decay habitat may develop over shorter time periods such as small diameter deadwood in the crown or bark shelters and may be fleeting in comparison to longstanding decay features such as heart rot (Courbaud et al., 2021). Nonetheless these more transient decay features still provide key resources to saproxylic species.



Left: Brown cubicle heart rot. Right: Diptera and invertebrate exit holes.

Figure 3: Left: Re-erecting fallen dead stems. Right: Conserving crumbling decay habitat using ratchet straps

Promoting the development of ancient and veteran tree microhabitats

Some practitioners have attempted to speed up the decay process in young trees and recreate the microhabitats associated with ancient and veteran trees in a process known as microhabitat creation, habitat pruning or veteranisation (Bengtsson et al., 2012).

The term veteranisation has led to confusion about whether such techniques are suitable for veteran trees. **Microhabitat creation should never be undertaken on ancient and veteran trees**. We therefore discourage the use of the term veteranisation in favour of microhabitat creation.

Microhabitat creation practices are diverse, including deliberate mechanical and cutting damage, inoculation of heartwood with heart rot fungi, beetle boxes replicating wood mould rich cavity habitats, and the creation of nesting boxes in live trees for mammals and birds. Many of these techniques show great promise for increasing the abundance of these critically threatened ancient and veteran tree associated microhabitats. However, we currently lack sufficient empirical data to support the widespread adoption of microhabitat creation. More research is needed, and better strategies for monitoring current work need to be developed to ensure its implementation is well informed and the desired conservation outcomes are secured.

Conserving deadwood habitat, *in-situ* (figure 3) where possible, is important for the active management of ancient and veteran trees. This could be as simple as leaving deadwood where it falls or translocating it to a more suitable area for retention if it cannot be retained *in-situ*. In some instances, interventions may be more complex; for example, using ratchet straps to maintain saproxylic habitat in the centre of old standing trees or even re-erecting fallen deadwood. The **size** and **heterogeneity** of deadwood alongside **microclimate** are important in influencing the diversity and composition of saproxylic communities (Stokland *et al.*, 2012). It's therefore important to promote a variety of deadwood habitat - from small branches to whole trunks - both standing and fallen on the ground, and with various levels of exposure to support our deadwood communities.

IMAGE CREDIT: REG HARRIS

Carved cavity

IMAGE CREDIT: REG HARRIS

Winch pruning or large limb

IMAGE CREDIT: ARON KIMBERLEE

Chainsaw carved bird box designed by Aron Kimberlee

IMAGE CREDIT: ARON KIMBERLEE

Chainsaw carved bird box. Design Aron Kimberlee

IMAGE CREDIT: LYNNE BODDY

Conservation inoculation of heartrot fungi.

Microhabitat creation is most appropriate on sites that already have ancient and veteran trees, but where there is a large age gap between them and the next generation of ancients and veterans.

Selecting suitable trees is critical. Microhabitat creation on living trees should only be applied to **young and mature trees of low ecological value**. These treatments are likely to be most beneficial when conducted close to already existing ancient and veteran trees providing colonisable habitat within dispersal distances of existing long-established decay habitats. Microhabitat creation can be integrated with other management practices; for example, ring barking of stems and branches can be used as an approach to halo thinning to modify light levels while maintaining saproxylic habitat.

Techniques such as cavity creation and winch pruning can be used to expose sapwood and heartwood and initiate decay processes. Specific features can be targeted for specific groups of organisms such as small mammals or birds.

In some specialist cases, inoculation of rare or threatened heart rot fungi may be used for conservation translocation but must be performed under expert guidance. This is not a common treatment and is rarely acceptable or appropriate, especially if detailed data about the species and the origin of strains used is unavailable.

Where are our ancient and veteran trees?

About the data

The Ancient Tree Inventory (ATI), held and managed by the Woodland Trust and openly available, is a data set of records of ancient, veteran and other notable trees in the UK, collected by citizen scientist volunteers and validated by specially trained volunteers to increase reliability.

As a continuously updated, opportunistically collected data set that consists of positive records only (no records of tree absences), interpretation comes with important caveats, perhaps most importantly that records are largely influenced by recorder effort and how this is unevenly spread geographically. Regardless, the ATI serves as a crucial resource for conservation efforts, helping to identify and protect these ecologically and culturally significant trees. By analysing the ATI, we can gain insight into the distribution, density and characteristics of these remarkable natural landmarks.

Number of trees in the ATI

As of August 2024, 233,201 ATI records of ancient, veteran and notable trees have been verified in the UK, with the vast majority of recorded trees existing within England (83%). Recent research has suggested that there may be 8-10 times more ancient and veteran trees in England then suggested by the current number of verified trees recorded in the ATI (Nolan et al., 2022). There are many unrecorded ancient and veteran trees waiting to be discovered and the lack of knowledge regarding their location and distribution is currently a barrier to securing better conservation outcomes for our most important trees.

While the ancient tree ecology of lowland England is famed for its large, majestic oaks and open wood pasture landscapes, ancient and veteran trees may possess greatly different appearances in different landscape contexts. For example, in upland and coastal Scotland, and other upland regions, where trees may face tough conditions and poor soils, they may be smaller in stature and windblown characteristics are common. Species like rowan, willows and downy birch frequently get blown over and develop new stems from the fallen tree, termed phoenix trees. These gnarled trees, which fall outside common perceptions of what an ancient tree looks like, may go unrecognised and hence be undervalued and under recorded.

Table 1. The total number of verified ancient and veteran tree records per country

Source: Ancient Tree Inventory

	England	N. Ireland	Scotland	Wales	UK total
Ancient tree	15538 (6.7%)	159 (0.1%)	2086 (0.9%)	1442 (0.6%)	19225 (8.2%)
Veteran tree	115390 (49.5%)	2658 (1.1%)	9179 (3.9%)	6835 (2.9%)	134062 (57.5%)
Notable tree	61774 (26.5%)	6798 (2.9%)	7079 (3.0%)	4263 (1.8%)	79914 (34.3%)
Total	192702 (82.6%)	9615 (4.1%)	18344 (7.9%)	12540 (5.4%)	233201 (100.0%)

Common trees in the ATI

Oak trees are the most frequently recorded species of ancient and veteran tree in the ATI (table 2). Over 2,300 species are known to be associated with oaks, making oak a keystone species (Mitchell et al., 2019) (Table 2). The unique conditions of oak heartwood, it's low pH and high content of inhibitory phenolic compounds, makes oak heartwood very durable and persistent, and therefore oak heart rot provides a high continuity of saproxylic habitat, potentially for several centuries. Beech is the second most recorded species of ancient and veteran tree in the ATI and while its ripewood (analogous to heartwood) is considered less durable than the heartwood of oak, it's still a haven for saproxylic biodiversity and its differing wood chemistry and decay characteristics lead to different deadwood communities. Important sites for ancient and veteran beech include Epping Forest and Burnham Beeches, which contain the highest numbers of beech. Other important former wood pasture sites for beech include the New Forest and Savernake Forest. Less common but important trees include species such as yew. These trees can be the longest lived of our old trees. The oldest tree in the UK is the Fortingall Yew (Perthshire, Scotland) which is estimated to be between 2,000 and 3,000 years old, although some say it is even older. The secret to the longevity of yew trees is their ability to fragment and regenerate young shoots. Many ancient yews, including the Fortingall Yew, exist as fragmented parts of what was once a single tree. The fragmented section can live on to act independently as if they were individual trees.

Table 2. The 15 most frequently recorded ancient or veteran trees by species or genus, which comprise 88% of all ATI records (as a % of all ATI records in brackets)

Source: Ancient Tree Inventory

Species	Ancient tree	Veteran tree	Notable tree	All records
Oak[a]	7601 (3.3%)	62547 (26.9%)	27226 (11.7%)	97374 (41.8%)
Beech	1267 (0.5%)	17194 (7.4%)	8583 (3.7%)	27044 (11.6%)
Ash	2093 (0.9%)	10229 (4.4%)	4687 (2.0%)	17009 (7.3%)
Lime[b]	567 (0.2%)	5127 (2.2%)	3375 (1.5%)	9069 (3.9%)
Yew[c]	1149 (0.5%)	2675 (1.1%)	5229 (2.2%)	9053 (3.9%)
Sweet chestnut	1107 (0.5%)	5534 (2.4%)	2373 (1.0%)	9014 (3.9%)
Sycamore	130 (0.1%)	2634 (1.1%)	3006 (1.3%)	5770 (2.5%)
Hawthorn	643 (0.3%)	3082 (1.3%)	964 (0.4%)	4689 (2.0%)
Scots pine	305 (0.1%)	2547 (1.1%)	1313 (0.6%)	4165 (1.8%)
Willow[d]	488 (0.2%)	2439 (1.0%)	1155 (0.5%)	4082 (1.8%)
Horse chestnut	24 (<0.1%)	1806 (0.8%)	2218 (1.0%)	4048 (1.7%)
Birch[e]	277 (0.1%)	1977 (0.8%)	1162 (0.5%)	3416 (1.5%)
Field maple	386 (0.2%)	2283 (1.0%)	704 (0.3%)	3373 (1.4%)
Alder	762 (0.3%)	1807 (0.8%)	747 (0.3%)	3316 (1.4%)
Hornbeam	766 (0.3%)	1981 (0.9%)	529 (0.2%)	3276 (1.4%)
All forms	17565 (7.5%)	123862 (53.2%)	63271 (27.2%)	204698 (87.9%)

a Includes trees recorded as "pedunculate oak", "sessile oak", and "hybrid sessile and English oak".
b Includes trees recorded as "common lime", "small leaved lime", "large leaved lime", "lime".
c Includes trees recorded as "Irish yew", "common yew", "yew" .
d Includes trees recorded as "white willow", "crack willow", "cricket bat willow", "bay willow", "grey willow", "goat willow" or "sallow willow".
e Includes trees recorded as "silver birch", "downy birch", "birch".

Size isn't everything

While ancient and other veteran trees are often large girthed in comparison to other trees of the same species, this isn't always the case (figure 4). While girth may be a proxy for age, the diverse climatic and soil conditions under which ancient and veteran trees grow, alongside factors such as competition, damage and past management, leads to high within-species variability in tree girths. For example, trees growing in harsh upland conditions with poor quality soils may be significantly smaller than their counterparts in more favourable and productive locations. Therefore, size relative to the species should not be the main or only criteria used to distinguish ancient and other veteran trees from mature trees. Some species such as oaks and sweet chestnut are examples of species that can grow exceptionally large. The largest recorded oak in the UK, the Marton Oak, has a remarkable girth of 14.02 metres. Other species never reach such great sizes, e.g.birch and hawthorn which have comparatively modest girths even in the ancient phase.

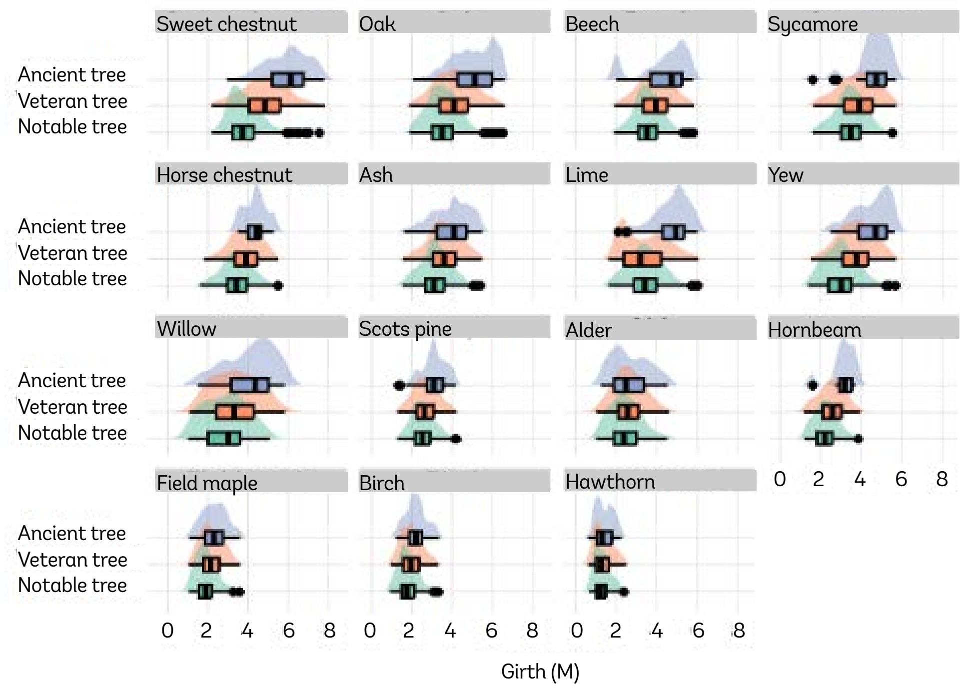


Figure 4 Box and ridge plots showing the distribution of girths for common tree species for ancient, veteran and notable trees in the ATI

Table 1. The total number of verified ancient and veteran tree records by tree form recorded in the Ancient Tree Inventory

Source: Ancient Tree Inventory

Veteran growth forms	Ancient tree	Veteran tree	Notable tree	All records
Maiden	7639 (3.3%)	77984 (33.4%)	61183 (26.2%)	146806 (63.0%)
Pollard	4582 (2.0%)	20173 (8.7%)	3158 (1.4%)	27913 (12.0%)
Multi stem	1629 (0.7%)	8025 (3.4%)	5700 (2.4%)	15354 (6.6%)
Unknown	773 (0.3%)	11148 (4.8%)	2956 (1.3%)	14877 (6.4%)
Coppice	773 (0.3%)	4224 (1.8%)	3536 (1.5%)	8533 (3.7%)
Pollard lapsed	1176 (0.5%)	3473 (1.5%)	679 (0.3%)	5328 (2.3%)
Pollard form (natural)	704 (0.3%)	2798 (1.2%)	1083 (0.5%)	4585 (2.0%)
Pollard managed	1025 (0.4%)	1458 (0.6%)	159 (0.1%)	2642 (1.1%)
Stump	281 (0.1%)	1813 (0.8%)	447 (0.2%)	2541 (1.1%)
Phoenix	152 (0.1%)	696 (0.3%)	375 (0.2%)	1223 (0.5%)
Multi stem (boundary)	187 (0.1%)	777 (0.3%)	146 (0.1%)	1110 (0.5%)
Coppice (high stump)	99 (<0.1%)	670 (0.3%)	232 (0.1%)	1001 (0.4%)
Stump (high >4m)	118 (0.1%)	493 (0.2%)	122 (0.1%)	733 (0.3%)
Laid (hedgerow)	75 (<0.1%)	272 (0.1%)	102 (<0.1%)	449 (0.2%)
Cliff tree	9 (<0.1%)	55 (<0.1%)	28 (<0.1%)	92 (<0.1%)
Hedgebank	3 (<0.1%)	2 (<0.1%)	8 (<0.1%)	13 (<0.1%)
All forms	19225 (8.2%)	134061 (57.5%)	79914 (34.3%)	233200 (100.0%)

Working trees

Many of the ancient and veteran trees recorded in the ATI have undergone some traditional management which highlights the value of these practices in the conservation of our oldest trees (figure 5). Trees that have undergone traditional practices, such as pollarding and coppicing, are sometimes referred to as working trees. These pruning practices involve the cyclical cutting and removal of stems and branches either at ground level or at several metres. Often pollarding and coppicing was performed to obtain products from the tree (e.g. fodder, firewood, charcoal and timber).

Pollarded trees have an undervalued cultural significance as remnants of historic land use systems, where the practice allowed the production of woody products while retaining land for grazing and crops. These heritage practices also perpetuate the lifespan of trees by promoting the rejuvenation of developmentally young shoots and stems. These old practices hold significant conservation value for deadwood habitats, and the species that depend on them. Pollarding and coppicing support the development of complex microhabitats, such as hollows and decaying wood, which are crucial for saproxylic fungi and invertebrates. The accelerated hollowing in pollarded trees fosters the continuity of these habitats over centuries, offering stable ecological niches for nationally endangered species, like the rare Moccas beetle *(Hypebaeus flavipes)*, which inhabits ancient oak pollards and is known from only one site - Moccas Park.

These previously common management practices across the whole of Europe have become largely abandoned for a variety of reasons, for example, economic unviability, devaluation in natural products and negative perceptions around pruning young healthy trees. Furthermore, many of our ancient working trees have been neglected; the weight of lapsed pollard stems, in combination with the significant decay habitat present in these trees means they are at risk of collapse. Restoring pollarding practices of young trees and careful reinstation of active management of lapsed pollards, could preserve these invaluable landscape features and ensure the survival of the biodiversity they support, maintaining long term habitat continuity for many specialist species.

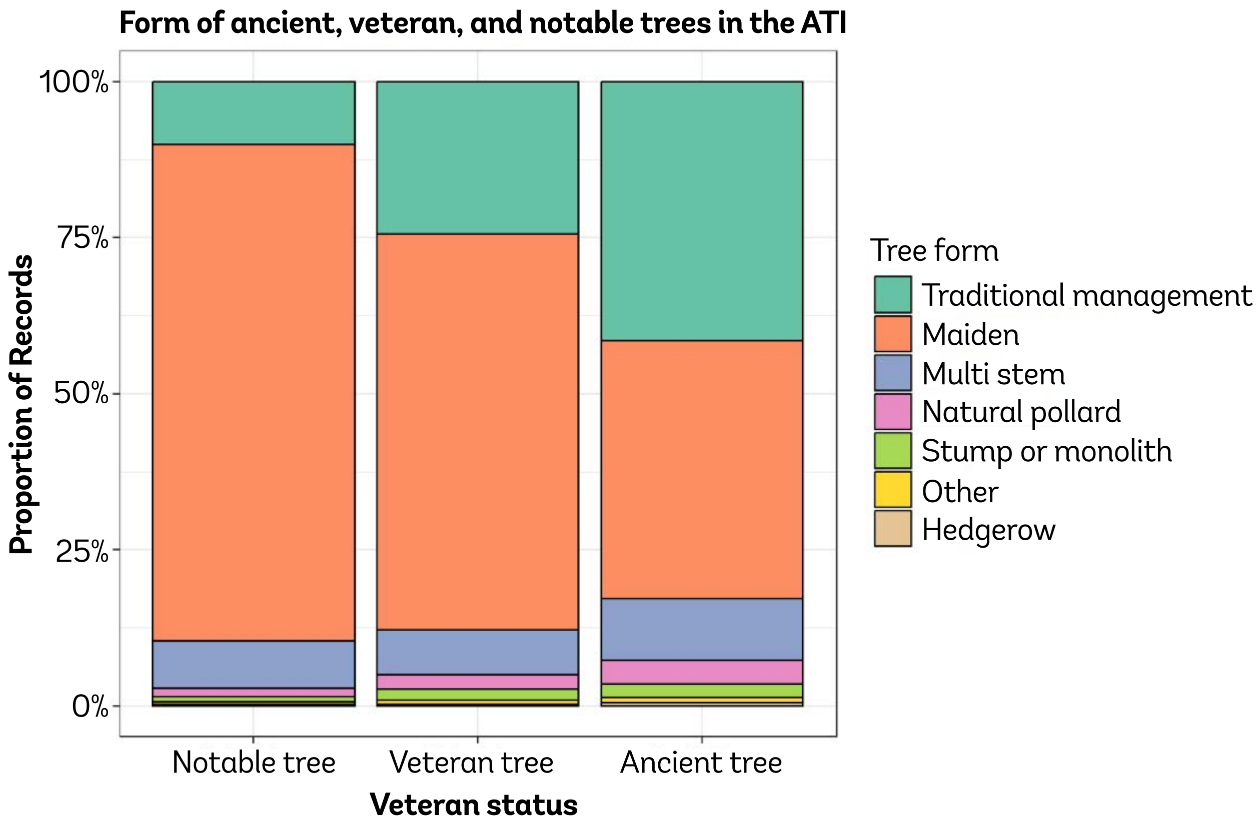


Figure 5: The proportion of trees that have undergone traditional management (pollarding or coppicing etc) increases with increasing veteran status.

Future directions and upscale of the ATI

While the ATI is a fantastic example of a long-lasting citizen science project with widespread engagement, it's opportunistic nature and recorder biases limits what we can reliably say about the extent and condition of our oldest and most special trees. Reducing our ability to take impactful conservation action.

The ATI is incomplete - A study utilising data from the ATI suggested that there are likely 8-10 times as many ancient and veteran trees in England than are currently recorded.

The ATI has geographic and recorder bias - A small number of recorders providing a large proportion of the records, geographical biases leading to under recording in areas.

The ATI is an opportunistic presence only database - Absence of records in the ATI should not be interpreted as an absence of ancient and veteran trees.

The ATI doesn't tell us how many trees we are losing (or recruiting) - Old age, poor management, land use intensification/change, development, pests and diseases, and climate change are all contributing to the decline of ancient and veteran trees and their associated habitats. We can't use the ATI to identify changes in demography of our ancient and veteran trees and drivers of this change.

The ATI doesn't track change – When records are revisited and updated in the ATI we do not currently have a method of recording these changes across time, which means we can't monitor the change in condition of our oldest trees.

What can we do about this?

To carry out effective conservation of our oldest and most special trees we need a better understanding of their extent, distribution and population trends to inform planning, decision making, and to identify conservation priorities. Distribution modelling based on opportunistic survey data, such as that of the ATI combined with independent and unbiased field survey validation is the gold standard for achieving this.

For example, in Norway, a national stratified sampling survey of veteran oaks allowed researchers to estimate that over periods of three to seven years, 7,600 veteran oaks are lost, resulting in an annual decline of 1.2% (Jacobsen et al. 2023). A UK study, using ATI data, combined species distribution model and field sampling to establish that there are 8-10 times as many ancient and veteran trees in England then the ATI alone currently suggests.

Using predictive modelling based on data from the ATI, a reduced bias stratified random sampling scheme has been developed for England and Scotland. For both nations the countries have been divided into 1km2 grids. Each grid was placed into one of five categories (strata) representing the predicted abundance of ancient and veteran trees within each grid. These strata were then used to identify 100 high-priority survey sites in each nation for a systematic ground-truthing survey. Unfortunately, work for Wales and Northern Ireland is lagging and we currently don't have strata predictions in order to identify high priority sites for sampling in these countries.

Currently these priority sites have not been surveyed. We now need to mobilise recorder effort efficiently to capitalise on this methodology. This could utilise contractors, tasked volunteer recorders, scheduled volunteer events and strategic alignment with existing projects to capture data. Alongside increasing dedicated sampling effort on the high priority sites, up-scale of the ATI would also require an increase in analytical capacity and the development of an analytical pipeline that allowed efficient incorporation of opportunistic and systematic survey efforts.

Development threats to ancient and veteran trees

In the following case studies, the fate of ancient and veteran trees highlights the tension between development projects and the conservation of our oldest and most special trees. While some trees were successfully protected, others were lost or still face significant threats due to development. These examples underscore the importance of robust planning policies, alongside their correct and appropriate interpretation and implementation, in order to preserve the irreplaceable biodiversity, cultural and heritage value of our most important trees.

Inappropriate interpretation of policy

The National Planning Policy Framework states that *"development resulting in the loss or deterioration of irreplaceable habitats (such as ancient woodland and ancient or veteran trees) should be refused, unless there are wholly exceptional reasons".* The phrase "wholly exceptional reasons" is intended to refer to nationally significant infrastructure projects, where the public benefit clearly outweighs the loss of habitat.

A local football club submitted a planning application for the conversion of their football pitch to a synthetic 3G surface, installation of new perimeter paths, fencing, floodlighting and a goal storage area. The development site was situated adjacent to a healthy veteran ash tree. The local tree officer expressed concerns that the proposed construction, although not directly involving the felling of the tree, would cause significant damage to the tree and its likely death.

Despite the tree officer's concerns and suggestions for alternative approaches, the football club rejected these options, in part due to disruptions to the football season. The planning committee ultimately voted to approve the application, contrary to the officer's recommendation, citing "wholly exceptional reasons" to justify the potential loss or deterioration of the veteran ash tree. These reasons included the community benefits related to the football club's viability and the new infrastructure's potential to enhance physical and mental health. Here, the interpretation of "wholly exceptional reasons" is tenuous and not aligned with national planning policy.

Ongoing conflicts, the Darwin Oak

In Shrewsbury, a controversial highways project threatens the destruction of nine veteran trees, including a 550-year-old, seven-metre girth oak known as the Darwin Oak. An additional 37 veteran trees have been identified as being at risk of deterioration. Despite clear conflicts with national planning policy, the local planning committee voted in favour of the project based on recommendations from the council's planning team. Although the committee's vote is yet to be ratified, this is a clear example of policy working against veteran trees.

While the decision has not yet been finalised, this case illustrates the challenges in enforcing protections for veteran trees when weighed against significant infrastructure projects.

Next steps

Apart from knowing gaps around where our ancient and veteran trees are, there are still large evidence gaps that limit our ability to campaign and influence policy and practice. We have very limited information on the condition of our most important trees, changes in population demographics and the rate of loss of our irreplaceable ancient trees.

Understand the threats: while we know our ancient and veteran trees face many threats, we lack data on the extent and magnitude of these threats. By better understanding threats and the management options we could implement to mitigate against them, we could increase resources and funding to target these threats more efficiently and implement well informed interventions at scale – this could involve developing specific funding sources to counter specific threats.

Bridge the gap: it is critical that we secure the next generation of ancient and veteran trees. We must ensure that our old trees are able to provide the spatial and temporal ecological continuity many of our most vulnerable species depend on. This could come from better stewardship of our ancients of the future through increased protection of our current cohort of mature trees. Greater exploration of microhabitat creation should also be explored, which may be able to bridge this gap, but not should be considered a replacement for naturally occurring microhabitats and is unlikely to truly replicate their ecological complexity.

Stronger legislative protection is needed for our oldest and most special trees.

Current legislative protection provided by **Tree Preservation Orders** (TPOs) and **Conservation Areas** do not adequately protect our most important trees and were not designed with the conservation of ancient and veteran trees in mind. The current legislation focuses on the amenity value of trees, while largely neglecting their ecological, cultural and heritage value. New legislative protection is needed that requires consistent levels of stewardship and protection wherever important trees are located. Our nationally important heritage assets like listed buildings and scheduled ancient monuments are legally protected and our important trees need similar protection.

For example, King John's Palace, in King's Clipstone, Sherwood Forest, is the remains of a royal palace or hunting lodge built around the mid-12th century. It is a scheduled monument for its historical and archaeological importance and its association with the highest echelons of medieval society. A mile or so away from the palace is the Parliament Oak – an ancient oak tree which dates from a similar time to the palace but has no similar type of heritage protection.

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Temperate rainforest

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Introduction

Temperate rainforests are globally rare habitats that occur in regions across the temperate zone where there are high levels of rainfall and oceanicity (the degree to which a region is subject to the influence of the oceans). Temperate rainforests are found along coastal and upland windward slopes where high humidity and low temperature fluctuations create conditions that are suitable for the growth of specialised plants and fungi, bryophytes, ferns and lichens. Temperate rainforests are characterised by a layer of epiphytes growing on, and within, the canopies of trees, and are unique and culturally significant woodlands (DellaSala et al., 2011; Shrubsole 2022).

Like tropical rainforests, these woodlands can support high numbers of highly specialist species and can provide significant benefits to human populations via ecosystem service provision (Brandt et al., 2014).

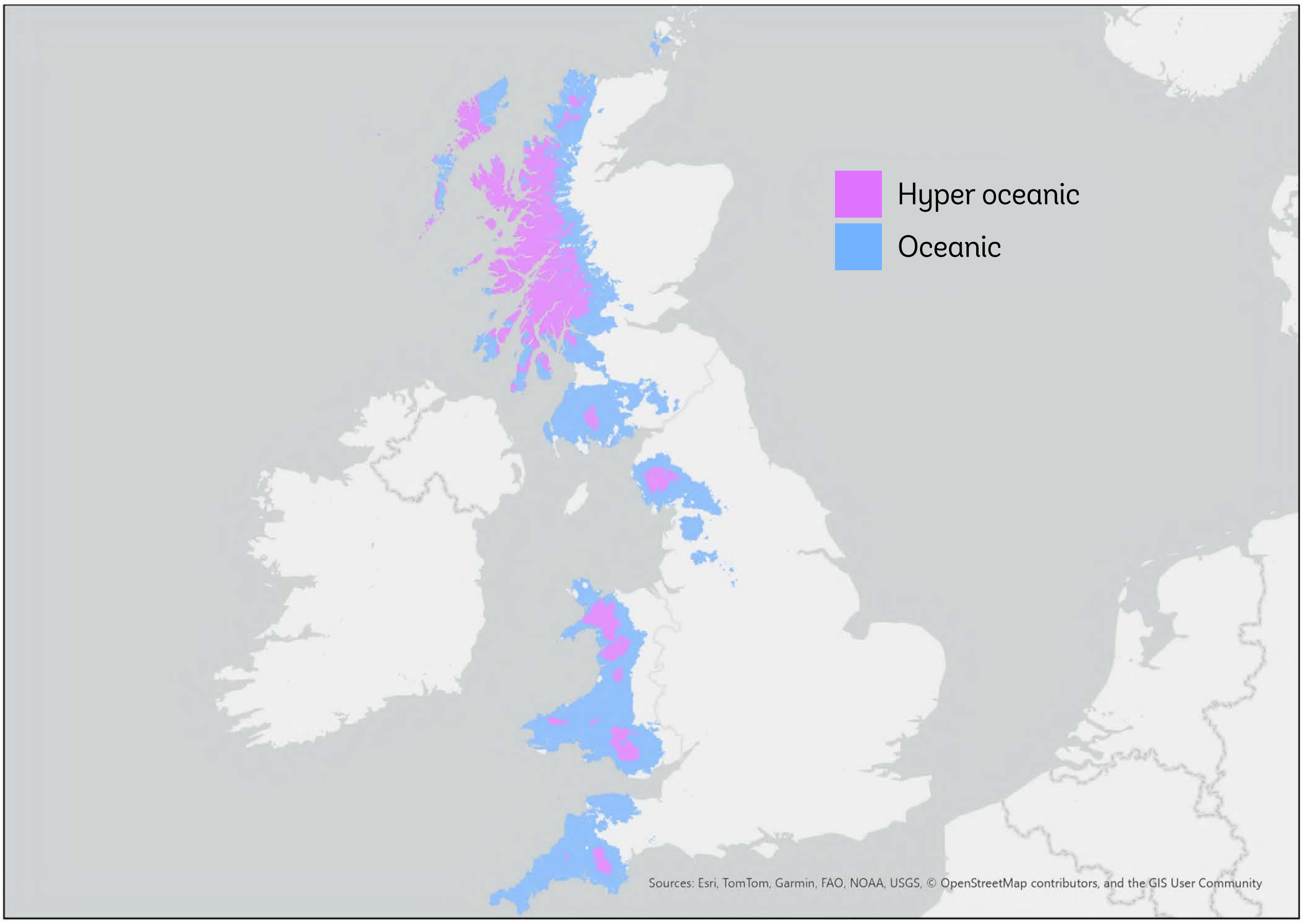
Temperate rainforests have the potential to store significant amounts of carbon both above and below ground and can influence regional water and potentially global climate dynamics (Murphy et al., 2020; DellaSala 2011; Zhu et al., 2015).

The global 'temperate rainforests' definition includes:

- > 1,400mm annual precipitation with at least 10% occurring in summer months.
- Cool, frequently overcast summers with a July mean < 16°C.
- Infrequent fires.
- Dormant season caused by low temperatures.

Based upon these criteria it has been estimated that conditions suitable for the formation of temperate rainforest occur across < 1% of the global land surface with 15% of this space occurring in Europe (DellaSala 2011).

The UK contains 40% of the current suitable climatic space for rainforest within Europe (DellaSala et al., 2011). The distribution of temperate rainforests across the UK is associated with regions of the greatest oceanicity, defined by hygrothermy values greater than 100 (Ellis, 2016). Using this index across the UK, 27.1% of land surface area occurs within an oceanic climate zone and 8.1% in hyper-oceanic conditions (Ellis, 2016).



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Temperate rainforests hold a deep cultural and historic significance; the loss of species and habitats is therefore not only a physical loss but also a symbolic loss.

Given the rarity and importance of temperate rainforests, it is vital we understand where they are, what condition they are in and what we can do to restore them. This chapter looks at our current knowledge based upon three State of Rainforest reports from Scotland, Wales and South West England.

Methods

Within the UK, the rainforest 'zones' are currently managed regionally (Scotland, Wales, South West England) through separate alliances, but the recent creation of the UK Rainforest Network aims to share lessons and knowledge between the regions over the coming years.

The results from the following reports were summarised to use in this section:

The Woodland Trust (2019). The state of Scotland's rainforests. https://www.woodlandtrust.org.uk/publications/2019/05/state-of-scotlandsrainforest/

Murphy T.R., Chernyuk, K., Roszkowski, M., Lewin, S., Maposa, N., Lunt, P.H., Buckley, J. (2024) State of Temperate Rainforest in SW England. Report and Mapping for Southwest Rainforest Alliance.

Alliance for Wales' Rainforest (2024) State of Wales Rainforests Report: The State of Wales' Rainforests Report | Celtic Rainforests Wales

Results

Northern Ireland

In Northern Ireland, the current situation with rainforest is different to the other three countries. Although the majority of the island of Ireland could be argued to fall inside the climatic conditions for rainforest, the term rainforest is not currently widely used. Instead, the rainforests of Ireland are predominantly in ancient woodland, and this is the focus of much of the restoration and protection work in Northern Ireland. For example, Mourne Park and the Faughan Valley have characteristics of rainforest and there is extensive restoration, buffering and connectivity work happening in these areas.

Due to the lack of terminology around rainforest currently in Northern Ireland, there is no data on the extent and condition of rainforest. Building on existing ancient woodland data, adding new information so that some of these can be categorised as rainforest, is vital in helping us understand where these precious habitats are and what is needed to restore them to a favourable condition. This could also help improve wider funding for rainforest protection and restoration, which currently has no direct funding stream for maintenance of woodland and instead comes in on a project-by-project basis.

Extent

- There are approximately 93,000 hectares of semi-natural woodland in Scotland's rainforest zone. However, only around **30,325ha (2% of Scotland's woodland cover)** can be classed as temperate rainforest due to its condition.
- The Welsh rainforest landscape covers an area of 768,000 hectares, which includes **116,042ha** of closed canopy woodland and forestry. This total is made up of 61,426ha of broadleaved woodland (53%) and 54,616ha of conifer plantation (47%).
- There is 878 km2 of existing woodland of rainforest potential in SW England (59 km2 is hyper oceanic, 819km2 is oceanic), 26% of which is mapped as ancient woodland, however the oceanic zone that could potentially support temperate rainforest covers almost 40% of SW England (9,597km^2).

Condition

- The Native Woodland Survey of Scotland has shown that only 30% (9,217ha) of Scotland's rainforest area is in satisfactory condition (Forestry Commission Scotland, 2014).
- There is not enough known about the condition of core rainforest sites in Wales. What little data is available shows that the habitat is in unfavourable condition. A Rapid Rainforest Assessment shows that only 22% of sites we surveyed are in good condition and 25% are in poor condition. No sites surveyed were in very good condition.
- Condition assessments of temperate rainforest in South West England are limited by the availability of consistent and up-to-date condition assessments. However, three independent sources of condition assessment suggest that the majority of temperate rainforest sites are in poor/ unfavourable condition. Of the legally protected woodland with temperate

rainforest potential in the South West, 55% are classified by Natural England as being in an unfavourable condition.

Threats

There are some common themes across the three reports around threats to temperate rainforests and how these need to be addressed including:

- **A lack of appropriate grazing management.** Grazing management is an important sustainable management tool for temperate rainforest. The grazing intensity needs to be at the right level. Too little grazing can lead to the growth of dense understoreys that threaten important rainforest species, while too much grazing can prevent regeneration and threaten the long-term survival, natural expansion and condition of the habitat. In Scotland, round 41% (c. 12,000ha) is suffering from high, or very high, levels of grazing, largely due to deer, impeding its long-term survival. In Wales 62% of sites surveyed have inappropriate grazing levels.
- **Invasive non-native species.** Rhododendron ponticum presents a significant threat to rainforest ecosystems as it creates dense shade which many rainforest species are unable to tolerate and prevents regeneration of species from the seed bank. In Scotland, rhododendron is present in at least 40% (12,290ha) of the rainforest area, and threatens overtaking the rainforest and reducing the distinctive rainforest flora. And in Wales, problematic invasive species, including Rhododendron ponticum as well as native species such as ivy and holly were present in almost 70% of sites surveyed.
- **Lack of woodland structural diversity.** The richness of temperate rainforests is linked to diversity; sites with diverse, complex structures tend to be much richer than those without. This is often linked to the continuity of ecological conditions on the site. Positive management can help tackle the lack of tree species diversity and structural complexity by opening up closed canopies and managing the amount of tree regeneration and invasive species in the understorey, and help with nature recovery.
- **Ash dieback.** Ash is one of the most important rainforest trees, both in terms of canopy cover and for the rainforest species it supports. This is especially so for the lichens that prefer its base-rich bark e.g. lungwort lichens which have populations of international significance in British rainforests. Climate change projections for the UK show milder, wetter conditions in which new pests and diseases could emerge and potentially thrive. Tree diseases are a current threat and are likely to be a constantly emerging threat to our rainforest e.g. acute oak decline and chronic oak decline. A total of 536 lichen species have been recorded from ash trees – as a staple feature of native woodlands, ash dieback poses a significant threat to Wales's lichen populations as we lose the tree that hosts so many epiphytic lichens.
- **Climate change.** The transition to warmer, drier summers and more erratic annual weather patterning because of climate change, is likely to result in a significant change to specialist rainforest species communities. A recent study has shown that unmitigated climate change would lead to a loss of 68% of the temperate rainforest biome. This could be limited to only 9% loss

with climate mitigation (Silver et al., 2024).

- **Habitat loss and fragmentation.** Due to human-driven land-use change and development, the rainforests of today occupy a vastly reduced area and now sit within a multitude of other land-use types. Rainforest specialist species are more likely to be resilient and able to adapt to climate change if they are less fragmented.
- **Plantation management.** Mid-20th century plantation establishment on sites of former native woodland – plantations on ancient woodland sites (PAWS)– have had negative impacts on rainforest extent and condition. These plantations can be composed of non-native conifers or broadleaves. While PAWS and long-established plantations can often still retain relict native rainforest habitat and species, they need careful management to protect and enhance this interest. Current restoration and management of PAWS is variable in its benefits for rainforest habitat and species. The most effective is a phased restoration based around halo thinning of surviving native trees that is adapted and designed for each site. The least effective is that based around clear-felling conifers leaving native trees isolated and vulnerable. A gradual restoration of PAWS is also needed in rainforest valleys where the conifers can help maintain high humidity levels. Clear-felling these trees can change the humid microclimate. In Scotland 21% (6,500ha) of rainforest area has been planted with non-native conifers.
- **Air pollutants,** such as sulphur dioxide, ammonia and nitrogen oxide, continue to have a huge impact on British habitats and species. The lichens and bryophytes that inhabit rainforests are particularly sensitive to atmospheric pollution, and because pollutants are often released in rainfall, the temperate rainforest landscape is particularly vulnerable to the effects of atmospheric pollution.

What's happening?

In Scotland, seven landscape-scale projects (covering just over 10% of the Scottish rainforest zone) have been set up to begin the restoration process. In 2021, the Scottish government committed to expand and restore rainforest and in 2024 allocated some funding to this. There are now 24 member organisations in the Alliance for Scotland's Rainforest. There are many projects looking at capacity building in rainforest communities and where we can use natural capital for restoration. The profile of rainforest in Scotland has risen significantly. Over the next few years, we hope to see the impact of this on the status of these precious woodlands.

Only 12% of Welsh rainforest has legal protection through the protected sites network as Sites of Special Scientific Interest (SSSIs). There are multiple landscape-scale projects happening across Wales to begin restoring the rainforest such as The Celtic Rainforest Wales LIFE Project and The Eryri Rainforest Partnership.

In South West England only 15% of temperate rainforest is legally protected by statutory designation (SSSI) with 3% of plantations on ancient woodland sites (PAWs) being statutory designated. Indicative restoration opportunity mapping highlights that 28.6% of the oceanic climate zone (or 11.5% of SW land area) or 51.0% of the hyper-oceanic climate zone (0.02% of SW land area) is highly suitable for temperate rainforest restoration and/or expansion. The

Woodland Trust/ South West Temperate Rainforest Alliance want to triple the area of temperate rainforest in the South West region by 2050.

Conclusion

Temperate rainforest is a unique and rare habitat that supports many globally rare and threatened species. However, temperate rainforest is also incredibly vulnerable to climate change with predictions showing that there could be a significant reduction in the climatic zone suitable for temperate rainforest. Other risks which are applicable to other types of woodland include appropriate levels of grazing, pests and diseases, pollution, invasive species and PAWS. This makes preserving the temperate rainforest we have more important than ever.

There is currently very little temperate rainforest under protection, and sensitive management and restoration is vital to ensure our rainforests can survive and thrive. Further data collection would help improve our understanding of the condition and necessary management needed.

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UK urban tree canopy cover and the urban forest

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Abstract

The 'urban forest'—comprising all the trees and woodlands within urban areas including grey belt land —provides vital ecosystem services such as shading and cooling, supporting physical and mental wellbeing, and stormwater management. These services help mitigate the effects of climate change and impact public health. Most of the UK's population lives in urban areas, where trees and woods make a vital difference to the quality of life for millions of people daily. However, expanding urbanisation is also a major cause of habitat loss, deterioration and fragmentation. Urban areas are also particularly sensitive to the effects of climate change, with elevated temperatures caused by the urban heat island effect contributing significantly to premature mortality in European cities. Despite the vital importance of the urban forest, information about levels of urban tree canopy cover (UTCC) across the UK is relatively scarce. Both published and anecdotal evidence suggest recent declines in UTCC across many urban areas. A lack of data around UTCC levels and trends can hinder the development of suitable policy to tackle widespread inequalities and develop baselines to monitor change. In this study, UTCC levels—for neighbourhoods, electoral wards and local authorities—are assessed on a UK-wide basis, determined from data provided by the Google Environmental Insight Explorer (EIE). The Google EIE dataset underpins the Woodland Trust's Tree Equity Score UK tool and uses recent known-provenance imagery and trained AI to interpret data. Analysis reveals wide variation between English regions and UK countries in terms of average UTCC. Currently 35% of UK urban electoral wards or 39.4% of local authorities have mean UTCC at, or above, a target of 20% previously set as an outline target for urban tree cover by the Government's Urban Forestry and Woodland Advisory Committee. Associations of neighbourhood UTCC with urban heat disparity data are also assessed, with a net cooling effect on summer surface temperatures seen at UTCCs above 20%. In addition to the above, a literature review is provided, detailing the benefits of the urban forest for ecosystem services and human health. The resilience and stress affecting the urban forest is also discussed, as well as contextual disservices of urban trees. This includes allergy to pollen, with increasing incidences of tree pollen allergy across the UK and Europe, partly due to earlier and greater volumes of pollen afforded by climate change. As well as accurately identifying levels of UTCC and targeting areas for improving the equitable provision of benefits of woods and trees in urban areas, urban planning for the future must consider the multiple competing objectives of urban trees and their contextual disservices.

Highlights

- Novel Google EIE derived dataset analysed for the first time on a UK-wide basis.
- Area-weighted mean UTCC value of 19.3% is determined for the UK, with

values of 15.2%, 18.8%, 19.5% and 22.2% for Northern Ireland, Scotland, England and Wales respectively.

- 35% of UK urban wards had a UTCC at or above 20%.
- Northern Ireland, North East England, the East Midlands, Yorkshire and the Humber, and North Wales have the lowest levels of UTCC.
- South East England and South Wales have the highest UTCC levels at every administrative level.
- Local authorities (LAs) with the lowest UTCC are Causeway Coast and Glens, City of London, Tendring, Great Yarmouth and Hartlepool, ranging from 3.5-7.7%.
- On average, UTCC levels above 20% lead to net cooling effect for UK neighbourhoods.
- Increasing neighbourhood UTCC from 10% to 20% would lead to summertime cooling of approximately 0.91°C compared to LA averages.
- UTCC is a useful index for establishing levels of canopy coverage, but not urban tree condition and diversity; an understanding of tree species, age and size are also needed to manage and protect the urban forest.
- The urban forest provides a range of other tangible benefits, including biodiversity, air pollution control and improved real estate prices; however, evidence is lacking in many areas.
- Urban cooling due to trees can directly reduce heat stress-related mortality.
- Birch pollen allergy is a growing issue across Europe and can have severe health outcomes including asthma. Tree pollen can also induce pollen food allergy syndrome, causing allergies to common foodstuffs.

Introduction

Increasing urbanisation is a long-term trend affected by population growth and economic development. More than half the global population now lives in urban areas, up from around one-third in 1950, and this number is projected to increase to around two-thirds by 2050 (UN, 2024). Around 83% of the UK population lives within urban areas (UN, 2018). Urbanisation can bring many economic and cultural advantages, but also a range of challenges, including higher living costs, housing shortages and crime (Zhang, 2016). Urbanisation also has profound environmental impacts, and urban expansion is a growing cause of habitat and species loss (Zhang, 2016; Li et al., 2022). At the same time, urban environments are also acutely sensitive to the effects of climate change, with the urban heat island (UHI) effect contributing significantly to premature mortality in European cities based on retrospective analysis (Iungman et al., 2023). Annual heat-related death rates for the UK and other Northern European countries are substantially lower than in Southern European populations (Iungman et al., 2023); however, this is expected to rise given an increased frequency of warm summers and extreme heatwave events, as well as lower baseline levels of urban tree coverage in the UK.

The urban forest comprises all the trees within urban areas and can help to mitigate the impacts of urbanisation and climate change through the provision of ecosystem services including biodiversity. Published and anecdotal evidence suggests that UTCC may be decreasing in many urban areas globally and within the UK (Natural Resources Wales, 2016; Doick et al., 2020; World Resources Institute, 2024). The Woodland Trust's Tree Equity Score UK tool (https://uk.treeequityscore.org), codeveloped with American Forests and the Centre for Sustainable Healthcare, has also highlighted widespread disparity in access to the benefits of urban trees across the UK, with some of the most affluent neighbourhoods having more than double the tree coverage of the least affluent (Figure 1). On average, neighbourhoods with more trees also have lower exposure to harmful air pollutants, cooler temperatures and better health outcomes. A Europe-wide UTCC target of 30% is identified by Iungman et al. 2023 to achieve reductions in UHI-related mortality, and result in greater climate resilience for cities.

Figure 1: Urban trees in a (a) Sheffield neighbourhood with a tree canopy cover of 23% and tree equity score (TES) of 100 (b) a neighbourhood in Grimsby with a tree canopy cover of 6% and a TES of 61

UTCC has been determined inconsistently in the UK, with a variety of assessments including the national map of Wales (NRW, 2016), the London Tree Cover Map (Greater London Authority, 2023) and Friends of the Earth/Terra Sulis map of England (Friends of the Earth, 2023). Several datasets are now available to assess UTCC at a city or national levels. However, each has relative strengths and weaknesses, and are not directly comparable (Doick et al., 2024). To indicate the variation between methods, post-2019 estimates of UTCC for the City of Birmingham have included values of 19.1% based on BlueSky National Tree Map data, 21.3% based on I-Tree Canopy data, 16.5% based on Environment Agency National LiDAR Programme data and 19.9% in the present study, based on data from the Google Environmental Insights Explorer (EIE) (Table 2; Figure 3; Treeconomics, 2023; Friends of the Earth, 2023).

To date, UK-wide estimates which have utilised a standard methodology remain limited to those derived by the Forest Research-led UK Ward Canopy Cover Map project which was reported on in the first State of the UK's Woods and Trees report (Reid et al., 2021). Starting with an estimate of UTCC in

283 English towns and cities (Doick et al., 2017), the map was developed over several years using human interpretation (including by Woodland Trust volunteers) of aerial imagery of unknown date derived from the publicly-available I-Tree Canopy tool (https://canopy.itreetools.org/), and has now provided canopy cover figures for every urban ward in the UK. Statistical analysis of these results has also recently been published (Sales et al., 2023). The average UTCC across the UK was found to be 17.3%, with Northern Ireland at 11.8%, Scotland at 15.7%, England at 17.5%, and Wales at 18.1%. Most electoral wards in the UK fall below a proposed target of 20% (Sales et al., 2023). Sales et al., 2023 also found a weak but significant negative association between UTCC and the index of multiple deprivation (IMD) across England, and Welsh and English wards also had negative associations between canopy cover and population density, meaning areas with higher population density were also likely to have lower tree cover (Sales et al., 2023). The UK Ward Canopy Cover Map project is now complete, and there are no plans to update this dataset at present (Forest Research, 2024). The lack of other UTCC estimates on a UK-wide basis hinders policy-making and resource allocation, and the development of a baseline for future monitoring.

The Woodland Trust's Tree Equity Score UK tool does not directly quantify UTCC itself. However, the tool uses a UTCC dataset generated by Google EIE. Google EIE provides recent (2021-2023) high resolution aerial RGB data which is passed through a human-trained segmentation model that categorises each pixel. Those pixels marked as 'tree' are aggregated to determine total canopy cover. The pixel sizes vary from 10cm to 25cm based on available data and may be from different time periods to obtain full coverage. For the Tree Equity Score UK Tool, UTCC is calculated at the same 'small area' statistical geography used by the national censuses and indices of deprivation in each UK country, using the EIE data. These are lower super output areas (LSOAs) in England and Wales, data zones in Scotland and super output areas in Northern Ireland (referred to as 'neighbourhoods' in this study). Through the Tree Equity Score UK tool, UTCC data has been generated for 34,299 UK urban neighbourhoods. The EIE data is constantly updated as new information becomes available from Google's sources, meaning it could be used to provide baseline estimates and ongoing assessment of change in UTCC across the UK.

In this analysis, UK-wide estimates of UTCC using the Google EIE dataset are provided for the first time, with breakdowns of area-weighted mean UTCC for UK nations and regions, electoral wards and local authorities. The association of neighbourhood UTCC with mean heat disparity data is also explored. Heat disparity is a measure used in the Tree Equity Score UK to compare average neighbourhood heat extremity with the local authority average to measure variance in heat severity across neighbourhoods. In addition, a literature review summarises the main benefits of urban trees and greenspace for ecosystem services, human health, ecological resilience and biodiversity. Contextual disservices of urban trees are also discussed.

Methods

Tree canopy cover

Recent (2022-2023) canopy cover statistics at a neighbourhood level (lower layer super output areas, datazones and super output areas) were provided by Google EIE as part of the Woodland Trust's Tree Equity Score (TES) Tool master dataset, created in partnership with American Forests and the Centre for Sustainable Healthcare and launched in December 2023. For England and Wales, LSOAs were selected for analysis if classified as urban major conurbation, urban minor conurbation, urban city and town, or urban city and town in a sparse setting. For Northern Ireland, urban SOAs were selected if classified as urban or mixed urban/rural. For Scotland, urban data zones were selected if classified as large urban areas or other urban areas. Urban and suburban land cover classes from the NERC EDS environmental information data centre land cover map 2021 25-metre raster were selected, isolated clusters smaller than 0.25 square kilometres discarded, and other land cover completely contained by the urban/suburban boundary reclassified as urban. A small number of neighbourhoods were excluded from the TES dataset; 527 LSOAs with missing data from across England/Wales, six English and 18 Welsh LSOAs were excluded based on complications adapting data to suit post-2011 boundary changes, and there were 18 missing datazones in Scotland. Neighbourhood data was then aggregated, and area-weighted mean values were determined for individual electoral wards, local authorities as well as UK nations and regions. For electoral ward and local authority data, arithmetic means were also calculated on a national and regional basis. Area-weighted mean UTCC was also determined for 11 selected cities across the UK, reflecting a variety of geographic and land use histories. As the raw remote-sensing pixel data used to inform UTCC data is not made available by Google, it was not possible to determine the standard error attached to statistics. Associations of UTCC with heat disparity scores were also assessed. Neighbourhood UTCC data were pooled to nearest whole value, and mean heat disparity calculated for each value. Heat disparity data was then plotted against UTCC values. For a full description of how surface temperature estimates were determined, see https://uk.treeequityscore.org. Heat disparity was calculated as:

$$TEMP_{diff} = TEMP_{n,ave} - TEMP_{la,ave}$$

where $TEMP_{n,ave}$ is the neighbourhood's urban area average of all maximum values for summer surface temperature from all Landsat 8 Collection 2 Level 2 scenes for the 2020-2023 period that intersect urban areas. $TEMP_{la,ave}$ is the local authority average of all maximum values for summer surface temperature for the same period. Thus, positive heat disparity values indicate hotter than average neighbourhoods in a local authority, and negative values indicate cooler than average neighbourhoods. Neighbourhoods set to the local authority average due to lack of data result in a heat disparity value of zero. All data analyses were provided by Treeconomics Ltd.

Literature review

A non-exhaustive literature review was conducted to provide updated information on the ecosystem service and human health benefits of urban forest/greenspace, as well as a discussion of resilience, biodiversity impacts and potential disservices of urban woods or trees. Searches primarily focused on UK studies but also included those from other regions or from a global perspective where necessary (for example global patterns in biodiversity loss). No date restrictions were used, but more recent research (post 2010) was prioritised. Searches for primary evidence were conducted using Google Scholar and included primary research as well as grey literature and websites of key resources or datasets. In combination with standard Boolean operators, Keyword search terms included amongst others: urban forest, UK, cities, street trees, ecosystem services, green space, particulate matter air pollution, climate change, pollen and resilience.

Results

Urban tree canopy cover descriptive statistics

Countries, regions and cities

Based on the Google EIE dataset, the area-weighted mean UTCC across the UK was 19.3% (Table 1), with figures of 15.2%, 18.8%, 19.5% and 22.2% for across Northern Ireland, Scotland, England and Wales respectively. Results were slightly higher than figures derived by Sales et al. 2023 (Table 1). The region with the highest UTCC was South-East England (25.5%) (Table 1; Figure 2) and the lowest in North-East England and Northern Ireland, each with mean UTCCs of 15.2% (Table 1; Figure 2). Wide differences in UTCC are apparent between and within cities, with neighbourhoods in Cardiff having the highest mean UTCC of the cities assessed, and Portsmouth the lowest (Table 2; Figure 2). Neighbourhoods in Greater London displayed the greatest overall range in UTCC, ranging 0-70% (Table 2).

Table 1: Urban Tree Canopy Cover (UTCC) for UK countries and regions determined from the Google EIE dataset, aggregated from results for LSOAs, Datazones and SOAs on a weighted mean basis

		Urban Area (km2)	UTCC (%)	N neighbourhoods	Sales et al. 2023
Total UK		31,210	19.3	34,349	17.3
Country	England	24,806	19.5	27973	17.5
	Northern Ireland	1,856	15.2	536	11.8
	Scotland	2,404	18.8	4597	15.7
	Wales	2,144	22.2	1243	18.1
Region	East Midlands	2,596	16.8	2127	15
	Eastern	3,096	18.8	2691	16.6
	London	1,525	20.8	4986	18.3
	North East	1,282	15.2	1375	16.3
	North Scotland	510	18.7	854	16.4
	North Wales	363	17.1	179	15.6
	North West	3,648	18.2	4121	15.9
	Northern Ireland	1,856	15.2	536	11.8
	South East	4,733	25.5	4452	22.1
	South Scotland	1,894	18.8	3743	15.6
	South Wales	1,781	23.2	1064	19.2
	South West	2,593	19.3	2376	15.7
	West Midlands	2,378	18.5	3044	17.4
	Yorkshire and the Humber	2,955	16.8	2801	17.3

Table 2: UTCC for 11 cities around the UK aggregated from results for LSOAs, Datazones and SOAs on a weighted mean basis, and range within constituent neighbourhoods

City	UTCC (%) (min-max)
Cardiff	23.2 (4.0-50.0)
Greater Manchester	20.8 (1.0-50.0)
Greater London	20.8 (0.0-70.0)
Plymouth	20.8 (4.0-48.0)
Edinburgh	20.1 (1.0-61.0)
Birmingham	19.9 (1.0-60.0)
Belfast	19.2 (3.0-41.0)
Glasgow	18.1 (1.0-54.0)
Leeds	17.4 (2.0-56.0)
Portsmouth	11.3 (2.0-26.0)

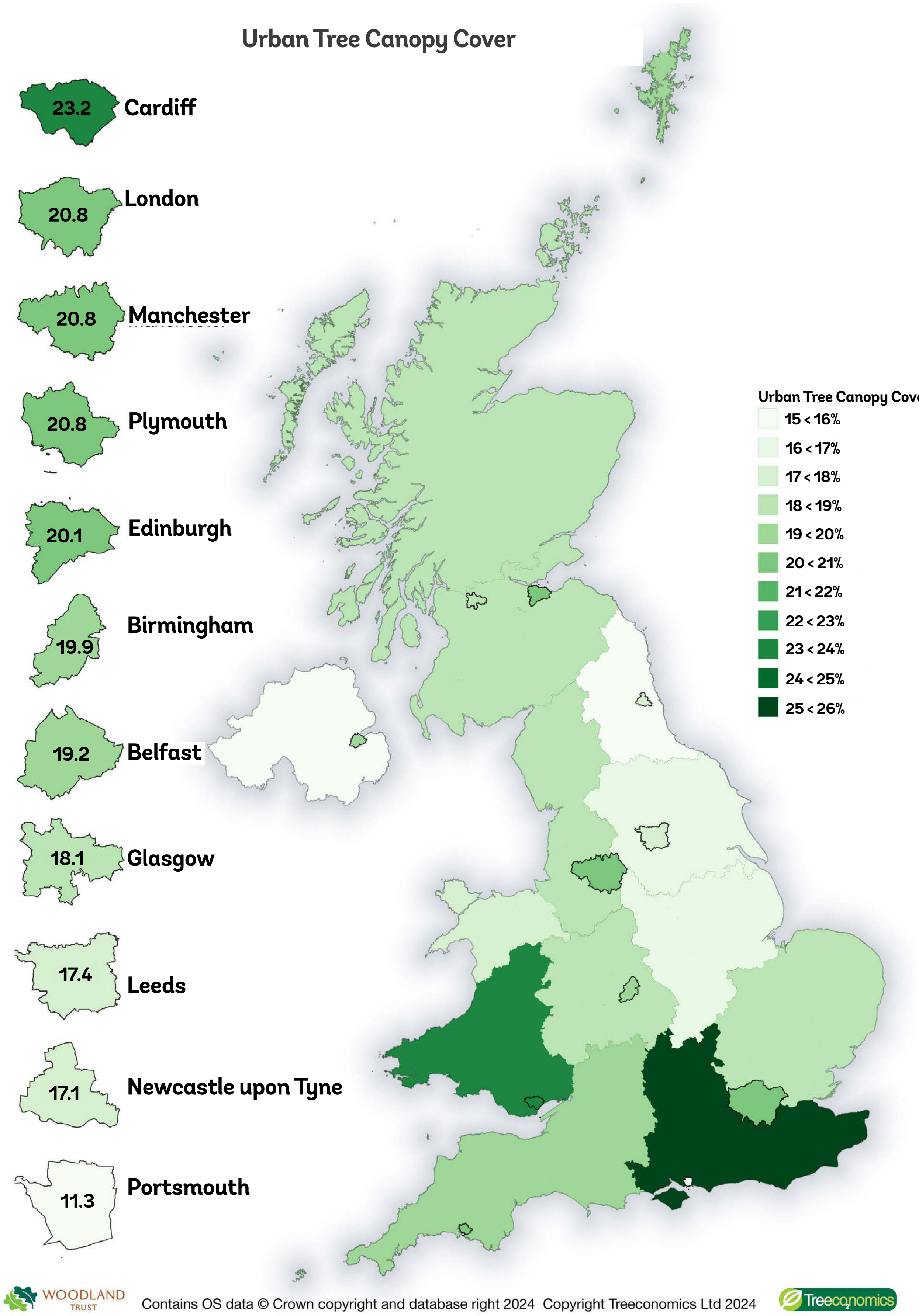

Figure 2: Area-weighted mean urban TCC for UK nations and regions, as well as 11 selected cities across the UK Electoral wards

Arithmetic mean UTCC across the UK was determined to 18.3% across 7,202 individual wards classified as urban (Table 3). Urban wards in Wales had the highest arithmetic mean UTCC (21%) and Northern Ireland the lowest (13.4%). Consistent with other estimates, wards in South East England had the highest mean UTCC, at 22.7%. South East England also had a widest range of values between electoral wards, ranging 3.1-58.6% (Table 3). Across the UK, the five electoral wards with the lowest UTCC were primarily found in East England, as well as Northern Ireland and the East Midlands (Table 4); conversely the five highest were all found in wards in South East England or London, ranging 52.5%-58.6% (Table 4). Of the 7,202 urban electoral wards assessed across the UK, 2544 (35%) had UTCC at or above 20% (see appendix).

Table 3: Arithmetic means, and min and max area-weighted mean UTCC values for electoral wards across UK nations and regions from the Google EIE dataset

		mean UTCC (%)	min	max	N wards
Total UK		18.3	Barms (<1.0)	Farnham Bourne (58.6)	7202
Country	England	18.4	Barms (<1.0)	Farnham Bourne (58.6)	6143
	Northern Ireland	13.4	Loughguile and Stranocum (2.0)	Aldergrove (46.3)	322
	Scotland	18.1	Kyle (3.7)	Fortissat (41.3)	244
	Wales	21.0	Rhyl West (2.3)	Chirk South (48.6)	493
Region	East Midlands	16.5	Barms (<1.0)	Valley (38.2)	655
	Eastern	18.4	Walton (1.0)	South Oxhey (46.0)	812
	London	17.8	Southall West (2.3)	Hampsted Town (56.0)	665
	North East	15.0	Coatham (2.7)	Ponteland West (36.0)	321
	North Scotland	17.8	Mid Formartine (8.0)	Lomand North (40.1)	60
	North Wales	17.7	Rhyl West (2.3)	Chirk South (48.6)	125
	North West	17.0	Peel (2.8)	Penrith North (40.4)	758
	Northern Ireland	13.4	Loughguile and Stranocum (2.0)	Aldergrove (46.3)	322
	South East	22.7	Stone and Waddesdon (3.1)	Farnham bourne (58.6)	1208
	South Scotland	18.2	Kyle (3.7)	Fortissat (41.3)	184
	South Wales	22.0	Tyisha (4.0)	Abercarn (47.5)	368
	South West	18.4	Portland (3.6)	Canford Cliffs (42.3)	656
	West Midlands	17.4	Churnet (4.3)	Cofton (45.1)	681
	Yorkshire and the Humber	16.6	East Marsh (4.4)	Monk Fryston & South Milford (33.3)	387

Table 4: UK electoral wards with the five lowest and highest area-weighted mean UTCC values

Country/ region	Local authority	Electoral ward	UTCC (%)
East Midlands	High Peak	Barms	<1.00
Eastern	Tendring	Walton	1.00
Eastern	South Norfolk	Easton	1.01
Eastern	Great Yarmouth	Nelson	1.69
Northern Ireland	Causeway Coast and Glens	Loughguile and Stranocum	2.04
South East	Waverley	Chiddingfold	52.5
South East	Bracknell Forest	Swinley Forest	55.0
London	Camden	Hampstead Town	56.0
South East	Chichester	Fernhurst	58.1
South East	Waverley	Farnham Bourne	58.6

Local authorities (LAs)

Arithmetic mean UTCC across LAs on a UK-wide basis was 19.2%; as with UTCC figures at neighbourhood and electoral ward levels, LAs in Wales had the highest values on average (21.0%) and Northern Ireland the lowest (14.0%) (Table 5). Regionally, the same patterns were also repeated, with LAs in South East England having the highest arithmetic mean UTCC (24.4) and broadest range (11.3-39.1)(Table 5). The LAs with the five lowest individual area-weighted mean UTCC values were Causeway Coast and Glens, City of London, Tendring, Great Yarmouth and Hartlepool, ranging from 3.5-7.7% (Table 6). The five highest were Sevenoaks, Bracknell Forest, Surrey Heath, Woking and Tandridge, all situated in the South East (Table 6). 39.4% (143) of UK LAs had UTCC at or above 20%, higher than the percentage estimated by Sales et al., 2023 (22.5%). Based on the England-only Terra Sulis/Friends of the Earth Tree Canopy dataset, just 14% (45) of English LAs had UTCC values $^{\geq}$20%, compared to 39.28% (121) in the present study (Friends of the Earth, 2023; see appendix).

Table 5: Arithmetic means, and min and max area-weighted mean UTCC values for local authorities across UK nations and regions, derived from the Google EIE dataset

		UTCC (%)	min	max	N LAs
Total UK		19.2	Causeway Coast and Glens (3.5)	Tandridge (39.1)	363
Country	England	19.3	City of London (4.5)	Tandridge (39.1)	308
	Northern Ireland	14.0	Causeway Coast and Glens (3.5)	Antrim and Newtownabbey (24.1)	10
	Scotland	17.7	East Ayrshire (11.0)	West Dunbartonshire (24.5)	25
	Wales	21.0	Isle of Anglesey (8.8%)	Caerphilly (27.5)	20
Region	East Midlands	16.8	Boston (10.7)	Derbyshire Dales (26)	35
	Eastern	18.8	Tendring (6.6)	Three Rivers (33.3)	45
	London	19.6	City of London (4.5)	Bromley (30.1)	33
	North East	14.7	Hartlepool (7.7)	Northumberland (18.2)	12
	North Scotland	17.8	Aberdeenshire (12.0)	West Dunbartonshire (24.5)	8
	North Wales	16.1	Isle of Anglesey (8.8)	Gwynedd (26.0)	6
	North West	17.8	Barrow-in-furness (11.0)	Eden (33.3)	39
	Northern Ireland	14.0	Causeway Coast and Glens (3.5)	Antrim and Newtownabbey (24.1)	10
	South East	24.4	Portsmouth (11.3)	Tandridge (39.1)	64
	South Scotland	17.6	East Aryshire (11.0)	East Dunbartonshire (21.5)	17
	South Wales	23.1	Vale of Glamorgan (17.1)	Caerphilly (27.5)	14
	South West	19.0	Sedgemoor (9.5)	West Devon (26.2)	29
	West Midlands	18.2	Staffordshire Moorlands (13.0)	Redditch (26.1)	30
	Yorkshire and the Humber	16.2	Hambleton (8.6)	Richmondshire (28.0)	21

Table 6: Local authorities with the five lowest and highest area-weighted mean UTCC values across the UK derived from the Google EIE dataset. Data from Sales et al., 2023 based on I-Tree canopy data as well as the England-only LiDAR-based Friends of the Earth/Terra Sulis dataset is also presented for comparison

Country	Region	Local authority	UTCC (%)	Sales et al. 2023	FoE/ Terra Sulis
Northern Ireland	Northern Ireland	Causeway Coast and Glens	3.5	10.9	n/a
England	London	City of London	4.5	3.8	4.7
England	Eastern	Tendring	6.6	8.3	8.2
England	Eastern	Great Yarmouth	7.4	7.3	9.0
England	North East	Hartlepool	7.7	7.0	5.7
England	South East	Sevenoaks	35.7	30.7	25.2
England	South East	Bracknell Forest	37.0	34.4	35.4
England	South East	Surrey Heath	37.4	42.4	36.1
England	South East	Woking	40.0	27.3	30.8
England	South East	Tandridge	39.1	33.0	22.2

Heat disparity

Analysis of mean heat disparity with pooled neighbourhood UTCC revealed a strong negative association, with neighbourhood UTCCs >20% associated with a net cooling effect in summertime temperature compared to local authority averages (Figure 3). At UTCCs below 20%, neighbourhoods tend to be hotter than local authority averages (Figure 3).

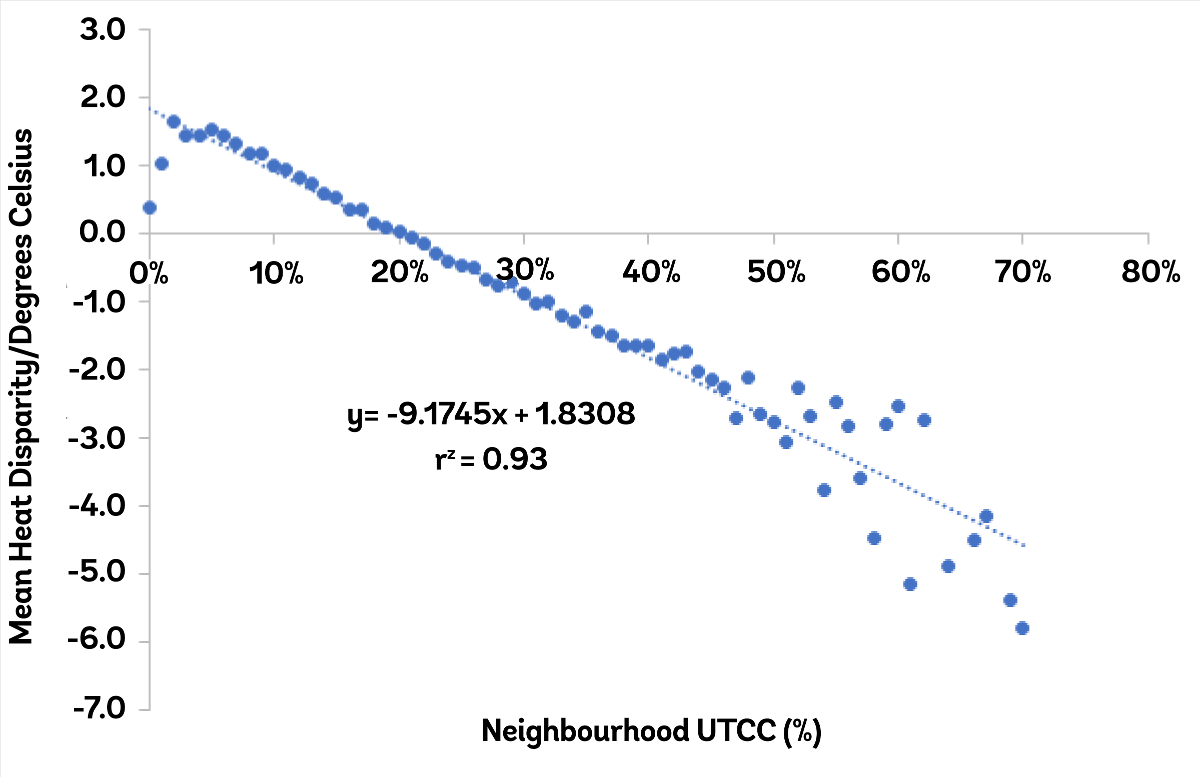

Figure 3: Mean summer surface heat disparity (difference between neighbourhood and local authority averages) across UK neighbourhoods, aggregated by UTCC values

Literature review – the urban forest

Ecosystem service benefits of urban trees and woods

Urban trees and woods can provide multiple ecosystem services (for reviews see Roy et al., 2012; Salmond et al., 2016; Pataki et al., 2021), the benefits of which are increasingly recognised by policy makers and local authorities (e.g. the London Urban Forest Plan (Mayor of London, 2020)). Positive services include (amongst others) beautification, increased real estate prices, stormwater absorption, carbon sequestration, noise attenuation, social cohesion and increasing pedestrian footfall. Trees, shrubs and hedges can also play a limited role managing exposure to polluted air in urban environments, acting as barriers to disperse or concentrate polluted air masses (Air Quality Expert Group, 2018). Although the effect is minor at a city-wide level, urban vegetation can also reduce air pollution levels; directly absorbing or capturing pollutants such as particulate matter (PM) and Ozone (O_3). Collectively, UK urban woods are estimated to remove 7.5 kilotons of pollutants per year, worth over £75 million in avoided health impacts (Jones et al., 2019). The value of ecosystem services provided by trees outside of woodlands across the UK – including urban trees – ranges £1.39-3.83 billion (Doick et al., 2022). From a climate change perspective, urban trees are thought to have a much larger role in adaptation than mitigation (Pataki et al., 2021). Trees – and other green infrastructure - can moderate urban microclimates, reducing day and nighttime surface temperatures via shading and evaporative cooling (Wang et al., 2018; de Quadros and Mizgier, 2023), which can increase thermal comfort of pedestrians (de Quadros and Mizgier, 2023) and reduce energy costs for buildings (e.g. Millward et al., 2014). Sweet chestnut (***Castanea sativa***), London plane (***Platanus hybrida***), sweet cherry (***Prunus avium***), sessile oak (***Quercus petraea***) and common beech (***Fagus sylvatica)*** have a high capacity to provide urban cooling (Moss et al., 2019). The ecosystem services of urban soils are under researched but serve a variety of functions including nutrient cycling and filtering and carbon storage (O'Riordan et al., 2021). Valuations of costs and benefits associated with urban trees are still developing and vary widely (Song et al., 2018). However, in general, the value of carbon sequestration, shading and air pollution abatement services provided typically breaks even or outweighs the cost of appropriate planting and maintenance, providing a lasting return-on-investment (ROI) as trees mature (Song et al., 2018). The value of some ecosystem services - such as biodiversity, noise reduction, tourism or aesthetics - have been inconsistently or barely determined but could have significant value (Song et al., 2018). For maximum ROI, the focus should be on preserving larger mature trees, as well as fewer but higher quality plantings with sufficient soil and root space for the developing tree (GreenBlue Urban, 2018). Growing evidence also supports a positive role of urban green space for human health (see below).

Urban greenspace and human health

Exposure to urban trees and greenspace such as parks is positively associated with multiple health benefits, including (but not limited to): reduced incidence of lung cancer, reduced incidence of childhood asthma and hospitalisation rate in periods of high air pollution, improved immune function and inflammatory markers, improved cardiac function, reduced exposure to ultraviolet radiation, lower rates of heatstroke and temperature-related conditions, improvements in cognitive ability, mood, sleep and stress recovery, reduced incidence of depression and increased birth weight (Dadvand et al., 2016; Gruebner et al., 2017; Astell-Burt and Feng, 2019; Yin, 2019; Astell-Burt et al. 2020; Wolf et al., 2020; Saraev et al., 2021; Maury-Mora et al., 2022). Longitudinal studies also indicate a directly positive effect of increasing UTCC on health markers; for example, lower incidence of dementia at higher UTCC rates (Astell-Burt et al., 2020)

In all cases, more research is needed to expand on and support these findings, particularly from a clinical perspective with controlled groups and larger sample sizes. The positive associations of greenspace with some health markers - such as cardiovascular, birth weight and overall mortality- are strongest for urban areas rather than in less urban and rural areas; however, greenspace exposure promotes other health markers equally across urban-rural gradients (i.e. obesity, respiratory and general physical health) (Browning et al., 2022). Existing socio-economic and health status likely modify the response of individuals to greenspace. In socially deprived neighbourhoods, proximity to greenspace can exacerbate mental health disorders and emergency hospital visits (perhaps due to anxiety associated with parks etc/wooded areas) (Yoo et al., 2022). Associations of greenspace with foetal growth may also vary by ethnic or cultural factors; for example, birth weights of Pakistani-origin residents in Bradford show no relationship to surrounding greenness in contrast to white British residents (Dadvand et al., 2014). Thus, proximity to greenspace does not necessarily mean equal access or outcomes. The health benefits of trees and greenspace are increasingly reflected in healthcare policy; for example, in the NHS Healthy New Towns Initiative (NHS, 2019) and the NHS Forest Network (Centre for Sustainable Healthcare, 2024).

Despite the many benefits of urban greenspace for health, the allergenic pollen of trees can also have a strong negative impact, and the consequences of allergic rhinitis can reduce outdoor activity, sleep quality, emotional wellbeing, and work/school performance (Bousquet et al., 2013; Léger et al., 2006; Canis et al., 2010). Trees within the Betulaceae (birch, alder, hazelnut, and hornbeam) and Fagaceae (oak, chestnut, and beech) families are thought to be the most important allergenic species in Northern Europe, forming the birch homologous allergenic group (Lorenz et al., 2009). Birch (*Betula* spp.) pollen volumes have increased significantly over recent years due to increased popularity as decorative species and an increase in pollen production caused by warmer temperatures (e.g. Frei and Gassner, 2008). Evidence suggests that the chronic stress caused by urban environments can also increase the volume and allergenicity of the most problematic birch pollen protein (Bet v1) (Beck et al., 2013; Stawoska et al., 2023). Thus, the problematic nature of tree pollen couples synergistically with other air pollutants such as PM and $O_{3,}$ as well as others (Beck et al., 2013; Stawoska et al., 2023). Consequently, sensitisation to birch pollen protein has increased across Europe, ranging from 8-16% in generalised populations (Biedermann et al., 2019). Symptoms associated with primary birch pollen allergy include rhinorrhoea, sneezing and congestion, as well as eye symptoms such as watering and redness. Birch pollen is also a major cause of asthma, and pollen deposited in the upper airways has the potential to cause a ripple effect throughout the whole respiratory system (Nevrlka et al., 2022; Gherasim et al., 2023). Birch pollen proteins have wide overlap, and sensitisation can result in cross-reactivity of antibodies to other tree species, which effectively extends the birch allergy season as other trees flower (Biedermann et al., 2019). Sensitisation to tree pollen can also induce some sensitivity to common food stuffs such as apple, peach and carrot, causing pollen food allergy syndrome (PFAS) in about 2% of the UK adult population (Allergy UK, 2021) and in 24-48% of children sensitised to pollen assessed across European populations (Mastrorelli et al., 2019). Symptoms of PFAS are typically mild local reactions such as tingling and itching. However, in about 2-10% of cases, mild symptoms can extend to systemic conditions such as throat swelling and hypotension. Life-threatening anaphylaxis affects about 1% of cases (Mastrorelli et al., 2019). Some strategies to reduce the extent of tree pollen allergy include prioritisation of low pollen species where children are at risk of sensitisation (i.e. schools and parks), use of female trees, increasing evidence and research to inform urban planning, and an increased diversity of planted species (Allergy UK, 2021)

Resilience, stress and biodiversity

Few studies have directly compared the environmental resilience of native trees or exotics, but the urban environment is often assumed to be less hospitable to native species. This is primarily due to the physiological traits of native species, which generally lack characteristics which are beneficial to dealing with chronic air pollution and drought such as a thick leaf cuticle (Grote et al., 2016). There is wide variation in physical traits within genera and within cultivars of the same species (e.g. in maple (*Acer*), Sjöman et al., 2015) which is largely unquantified. Street trees contend with a broad range of stressors including high temperatures, soil compaction, nutrient deficiencies and vandalism (Czaja et al., 2020). Water stress is a common factor exacerbating susceptibility of temperate trees to disease and temperature change, and improved hydration of urban trees could substantially reduce the compounding effects of threats (Meineke and Frank, 2018). Species choice often requires potential trade-offs depending on objectives and stakeholders. For example, a species that is more resilient to urban stress may be allergenic. Within the literature it is widely accepted that there is a pragmatic need for both native and non-native street trees in meeting the many objectives of urban greenspace (e.g. Sjöman et al., 2016).

Urban trees play a key role preserving or enhancing biodiversity in urban areas. However significant knowledge gaps remain, including for individual species and taxa, the role of urban microhabitats, long term biodiversity trends, human-wildlife interactions and the effects of climate change (Alvey, 2006; Gill et al., 2007; McDonnell and Hahs, 2013; Soulsbury and White, 2015). Research from around the world indicates that native trees generally support more urban biodiversity than non-native exotics (Southwood, 1961; Burghardt et al., 2010; Helden et al., 2012; Narango et al., 2017), and the amount of remnant native vegetation is one of the most important predictors of plant extinction in urban areas (Hahs et al., 2009). Exotics can also pose a risk of invasiveness and increase homogeneity across urban landscapes. Urban trees and tree pits, yards, gardens and roadside verges can act as biodiversity hotspots and act as stepping stones for species (O'Sullivan et al., 2017; Lynch, 2018; Ossola et al., 2019; Lundquist et al., 2022). Together with woods in urban peripheries, these can enhance connectivity in the urban forest, preserving biodiversity and ecosystem function (Lynch et al., 2018). However, habitats and trees within the urban forest can also suffer significantly from isolation, with an increasing proportion of birds and wind-dispersed species as the distance from rural areas increases (e.g. Olejniczak et al., 2018). Mycorrhizal and microbial diversity is likely to be significantly lower for trees in urban environments compared to rural counterparts (Bainard et al., 2011). Readily available food or habitat in urban areas can boost the population of many animal species, including rodents, pollinators, foxes and gulls and other birds. Biodiversity-friendly planting schemes should aim to maximise the proportion of native species (Liu & Slik, 2022). The importance of urban trees and greenspace for restoring nature to urban cities is recognised across UK governmental policy or initiatives; for example, the Scottish Government's green infrastructure strategic intervention (GISI) project (JNCC, 2021). In England, the Government has recently published a green infrastructure framework consisting of five GI standards to guide local policymaking, including a standard for urban nature recovery; this requires the locally determined expansion of nature-focused GI and restoration of urban wildlife-rich habitat (Natural England, 2023).

Disservices of urban trees

It is important to acknowledge the potential disservices of urban trees. This includes other health and safety issues such as trip hazards and exposure to urticating hairs from oak processionary moth (*Thaumetopoea processionea*) infestations, high tree mortality, complaints about leaf, branch or fruit litter, light attenuation, root or branch damage, increased gentrification, concerns about invasive species and pests, and wildfire risks (Roy et al., 2012; Townsend and Guest, 2018; Roman et al., 2021; Donovan et al., 2021; Pataki et al., 2021; Shi et al., 2023). Trees can also exacerbate air pollution by acting as a barrier to air movement or via natural emission of volatile organic compounds, which act as precursors of downwind particulate matter and O_3 (AQEG, 2018). Shading of buildings can increase energy demand on a site-by-site basis (e.g. McPherson et al.,1988). Disservices of urban trees can be reduced with appropriate establishment and aftercare. Urban forest research, policy and management should take an integrated approach to trees and consider both the potential positive and negative aspects through life-cycle assessment, analysis of ecosystem services and ROI analysis (Roman et al., 2021).

Discussion

Descriptive statistics from the novel Google EIE dataset revealed wide variation in patterns of UTCC across the UK, and results are broadly aligned with earlier analyses despite differences in methodology. This includes a UTCC figure of ~20% for Greater London (Treeconomics, 2015), the high UTCC levels of LAs in South East England, such as Bracknall, Waverley and Tandridge (Table 6; Sales et al., 2023) and the divide in canopy cover between North and South Wales towns (NRW, 2016). The low UTCC of neighbourhoods in many coastal cities or towns such as Portsmouth, Great Yarmouth, Clacton-on-Sea and Hartlepool is also noteworthy and supports calls for further investigation (Sales et al., 2023). In general, compared to LiDAR-based methods, the Google EIE dataset may slightly overestimate UTCC figures as height data is not included, meaning the boundary between shrubs and trees is unknown (Doick et al., 2023). In the present analysis, LSOAs, SOAs and datazones were included if classified as urban by UK or regional government definitions; thus, suburban land in the vicinity of neighbourhoods was also included to provide a more realistic approximation of urban spread and human activity, rather than definition based on built-up area or populations within each ward. Neighbourhood boundaries are also less likely to change than electoral wards or local authorities and allow easier comparison with census or other socio-demographic data. As Sales et al., 2023 note in their analysis, UTCC figures for UK cities are well below the European average (30.2%, EEA, 2021) and would require sustained long-term planting initiatives to reach targets of 20% (Doick et al., 2017) or 30% (Konijnendijk, 2022). Ambitious planting targets are widespread at a city level across the UK and in Europe; for example, London's environmental strategy goal of expanding UTCC by 10% by 2050 (Mayor of London, 2018) and the European Environment Agency goal of planting three billion trees by 2030 (EEA, 2021). While UTCC is a useful index for establishing levels of tree coverage, it cannot inform on urban tree condition or diversity; management of the urban forest also requires a good understanding of tree species, size and age classes. Associations of

UTCC with index of multiple deprivation were not directly assessed here, but a close association is known from the Tree Equity Score UK tool. The strong negative association found between neighbourhood UTCC and heat disparity reinforces the importance of urban trees in moderating temperature, with a summertime net cooling effect relative to local authority averages seen in neighbourhoods with UTCCs above 20%, below the 30% UTCC target identified by Iungman et al., 2023. Increasing neighbourhood UTCC from 10% to 20% would lead to a cooling of approximately 0.91°C compared to LA averages. However, some positive improvement in public health could occur with even modest increases in neighbourhood planting levels. Preserving and increasing UTCC is thus essential to obtain the tangible benefits that can be provided by the urban forest, and particularly to help mitigate climate change effects. There remains a lack of urban biodiversity monitoring, and a straightforward relationship between increasing UTCC and urban species richness cannot be assumed; particularly where native species are not prioritised. It should also be recognised that trees are not a panacea to the environmental and societal issues facing urban centres: presenting a range of contextual disservices and requiring suitable placement and maintenance to adapt to multiple chronic stress factors.

Urban forests provide a host of ecosystem services and can play a key role in improving the climate resilience of cities, including reducing heat-related mortality caused by the UHI effect. However, analysis of the UK-wide Google EIE dataset suggests that urban tree canopy cover figures for the UK are well below the European average, and significant variation exists at every administrative level. Despite the many positives of urban trees, planting policy and urban forest management must also take into account their potential disservices - particularly rising levels of tree pollen allergy.

Evidence gaps

- Research into the historical/current reasons behind low UTCC levels in coastal cities/towns
- Biodiversity benefits of the urban forest/green space, and challenges facing biodiversity
- Quantifying the effect of urban trees and vegetation on air pollutant removal
- Continued research into associations of urban trees/green space with physical/mental health outcomes

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Access and wellbeing

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Abstract

Poor health and wellbeing can have a profound impact on individuals, communities and wider society. Non-communicable diseases, including poor mental health and wellbeing, present the greatest global public health challenge in the 21st century, including here in the UK. Tragically, there is stark socioeconomic inequality in the health and wellbeing of the UK population, with the heaviest burden of poor health and wellbeing experienced by socioeconomically deprived communities.

There is consensus that spending time in nature can decrease the risk and burden of poor health, and elevate people's wellbeing, leading to considerable savings to the health system. Publicly accessible natural spaces are therefore vital infrastructure for supporting the population's health and wellbeing.

This report section draws together the most recent evidence around the role UK woodlands play in delivering wellbeing benefits for people. It presents evidence on:

- How much economic value the health benefits derived from visiting woodlands contributes to the UK economy, based on estimates calculated by Forest Research and the Office for National Statistics.
- How many people regularly visit woodlands, using UK-wide data from Forest Research's Public Opinions of Forestry survey,
- What prevents more people from regularly visiting woodlands, and how these barriers could be overcome, based on the findings of a literature review by Forest Research.
- What we know (and what we don't know) about how many people have access to a woodland close to their home in the four countries of the UK, based on a summary of the available data in each country.
- What the barriers are for woodland managers which prevent more woodlands being made publicly accessible, and how these barriers could be overcome, based on the findings of a literature review by Forest Research.
- How the quality of woodlands influences their value for wellbeing, including the role of biodiversity for wellbeing experiences, based on a non-systematic review of the literature and findings from a recent Woodland Trust-funded research project.
- How wellbeing-biodiversity quality is distributed across the UK, and whether this distribution is socioeconomically equitable, based on a spatial analysis undertaken as part of the Woodland Trust-funded research project.

Key findings are that:

- The annual mental health benefits associated with visits to the UK's woodlands were estimated to be worth £185 million per year (at 2020 prices) (Saraev et al., 2021), and annual overall health benefits were valued at £1.149 billion in 2022 (Office for National Statistics, 2024).
- Many people in the UK (26%) report they have not visited a woodland "in the

last few years". Most people in the UK do not visit woodland regularly, with 34% of those who do visit woods, visiting more than once per month in the summer, and 23% visiting more than once per month in the winter (Forest Research, 2023).

- Some of the key barriers to woodland access for members of the public include the distance needed to travel to reach woodlands, the availability and cost of transport, poor physical health or disability, concerns about personal safety, a perception that woodlands are not inclusive places for a diverse range of people, lack of time and lack of interest (Pearson et al., 2023).
- Data on how many people meet the Woodland Access Standards has not been updated for Wales, Scotland or Northern Ireland since the previous State of the UK's Woods and Trees report. A new, more comprehensive analysis for England is being carried out by Forest Research and was due to be published in March 2025, but, at the time of writing, results are not yet available. That analysis will also test whether there is socioeconomic inequity in the distribution of public woodland access across England.
- Key concerns for woodland managers which may prevent more woodlands being made publicly accessible include: the cost of installing and maintaining infrastructure; poor visitor behaviour such as littering; disturbance to wildlife and livestock, especially from dogs; bureaucracy of grant management; and concerns around liability for the health and safety of the public on their land (McConnachie & Gardner, 2024).
- The quality of woodlands is important for wellbeing. This includes visitor infrastructure (Pearson et al., 2023), but also the structural (Beute et al., 2023) and ecological characteristics of the woodland, as sensory engagement with biodiversity in woodlands is important for wellbeing (Bentley et al., 2023; Fisher et al., 2023).
- New research funded by the Woodland Trust found that 90% of people report positive wellbeing responses to the biodiversity in their local woodlands (Fisher et al., 2024 preprint).
- The biodiversity-wellbeing quality of woodlands (the potential sensory experience offered by rich woodland biodiversity) is unequally distributed across the UK, with woodlands in the South East of England, and some areas of Scotland having particularly high potential to deliver wellbeing across all seasons. There is inequity in this distribution, as woodlands estimated to contain the lowest biodiversity-wellbeing quality are located in more socioeconomically deprived neighbourhoods. This is according to new research funded by the Woodland Trust (Fisher et al., 2024, preprint; Fisher, 2025a, manuscript in preparation).

There are many challenges to wellbeing in our rapidly changing world, from financial stress exacerbated by the current cost-of-living crisis to climate anxiety and social isolation. Supporting people from all parts of society to build a loving relationship with the rest of nature is more important than ever: it is essential for improving the wellbeing of individuals, and for shifting values which lay the foundation for the societal behaviour changes required to restore nature and climate. Population-level reconnection with nature can be supported through improving the level of access to, and facilitating increased engagement with, natural places including woodlands. It is also critical to recover woodland biodiversity at a landscape scale, so that degraded landscapes can once again support the range of species which contribute to the wellbeing experience of visiting rich and thriving woodlands.

Background

What do we mean by health and wellbeing?

The World Health Organisation (WHO) defines health as 'a state of complete, physical, mental and social wellbeing and not merely the absence of disease or infirmity' (WHO, 2024).

Human health and wellbeing in a changing world

Whilst humanity made significant progress over the previous century on improving human health and wellbeing, largely by reducing the threats posed by infectious diseases, the 21st century is witnessing the unfolding of what some describe as a 'pandemic' of noncommunicable diseases (Allen, 2017). Noncommunicable diseases (NCDs) are a group of conditions that include cardiovascular disease, chronic respiratory diseases, cancers, diabetes and mental illness. These diseases cause the majority of deaths in the UK and severely impact both life expectancy and healthy life expectancy, causing significant personal suffering and placing a huge demand on the NHS (Office for Health Improvement and Disparities, 2021). Globally the number of deaths from NCDs is rising rapidly and NCDs are now the leading cause of death in developed and developing countries alike (World Health Organisation, 2023).

Sometimes referred to as lifestyle diseases, a dominant narrative has been to view NCDs as the self-inflicted consequence of poor individual lifestyle choices. However, in recent years, it has become apparent that systemic social, political and economic changes are the most significant driving forces behind the rise of NCDs across the world (World Health Organisation, 2023), rather than a sudden global collapse in personal responsibility.

As with other NCDs, the costs allied with treating poor mental health are expanding worldwide (Rippe & Egger, 2024). Mental ill health is one of the greatest challenges facing healthcare in Europe, affecting 13% of the population (World Health Organisation, 2022) and mental health problems cost the UK economy at least £117.9 billion per year (McDaid et al., 2022). The prevalence of mental health disorders such as anxiety and depression are increasing (McManus et al., 2016). Growing public awareness and concern about the climate and ecological crisis is driving the relatively new phenomenon of eco-anxiety (Pihkala, P., 2020).

Wellbeing predicts mortality and morbidity (Steptoe, 2018). Poor mental health can be the first step underpinning behaviours which set in motion the development of a cascade of inter-related physical health conditions, highlighting the potential power of improving mental wellbeing for many positive subsequent health outcomes (Kivimäki et al., 2020).

There are severe health and wellbeing inequalities related to socioeconomic status in the UK

In many countries, including the UK, low socioeconomic status is strongly associated with increased risk of ill health, especially from NCDs (Kivimäki et al., 2020). A third of premature deaths in the UK can be attributed to socioeconomic inequality (Lewer et al., 2020). For example, in England, people in the most deprived socioeconomic group are four times as likely to die from cardiovascular disease as those in the least deprived socioeconomic group (Office for Health Improvement & Disparities, 2021).

The role of nature in helping maintain health and wellbeing in a changing world

People have long known that being outdoors and in nature is good for them. The public generally agree being in nature makes them happy (Natural England 2023). There is now consensus that time in nature elevates human health and wellbeing (Lovell et al., 2020), decreasing the risk and burden of some types of mental illness (Bratman et al., 2019a).

Nature-based recreational physical activity results in reduced burden of disease and considerable annual savings through prevention of NCDs. It's estimated that visits by England's public to natural environments in 2019 facilitated enough physical activity to prevent between £70 million - £150 million in avoided healthcare and societal costs of ill health (Grellier et al., 2024).

The covid-19 pandemic shone a spotlight on the importance of nature for health and wellbeing, with many people spending more time noticing nature, and valuing the importance of greenspace and wildlife for their wellbeing (Natural England 2022).

People who feel more connected to nature tend to experience better wellbeing (Pritchard et al., 2020). Beyond just spending time in nature, it appears that actively noticing nature is important for building nature connectedness (Richardson et al., 2022). This feeling of connection to nature also leads to more pro-environmental choices of behaviour (Martin et al., 2020).

Nature and human health and wellbeing are inextricably connected (World Health Organisation, 2021). It has been argued the crises of climate change, biodiversity loss, and human health and wellbeing are linked to a society-wide disconnect from the rest of nature (Richardson et al., 2020), and therefore that fixing this relationship with nature is critical to ensure our mutual flourishing into the future (Krashi et al., 2024).

Spending time experiencing nature is associated with reduced socioeconomic inequality in health and wellbeing.

Time spent in nature is associated with greater self-reported wellbeing and appears to reduce inequality in wellbeing by narrowing the difference reported by those living in socioeconomic deprivation, and those who do not (Abdallah et al., 2017; Garrett et al., 2023; R. J. Mitchell et al., 2015). This effect became even more apparent during the covid-19 pandemic (Geary et al., 2021).

Exposure to green environments is also associated with reduced risk of poor mental and physical health particularly for those living in more deprived neighbourhoods (Geary et al., 2023; R. Mitchell & Popham, 2008). This suggests improving access to nature could go some way to disrupting the conversion of socioeconomic inequality into inequality of health and wellbeing (R. J. Mitchell et al., 2015; R. Mitchell and Popham, 2008). People with lower socioeconomic status tend to experience the strongest health benefits from public green spaces and parks, perhaps because lack of access to other health-promoting resources leads to a greater dependency on local green space (Rigolon et al., 2021). In the context of the current cost-of-living crisis, when many people are facing intensified financial strain, this effect could have increased importance for supporting public health.

The positive outcomes from spending time in nature are of significant interest to the public health sector. The need for access to nature is increasingly recognised as part of the complex landscape of solutions to tackling poor health and health inequality (Thomson et al., 2021). Consequently, initiatives such as 'green social prescribing' now support people to engage in self-directed and organised nature-based activities for their health and wellbeing (Husk et al., 2020). For the nation to successfully embed a population-level increase in the use of nature as part of a strategy for tackling the burden of poor health and wellbeing, it is essential to understand the many factors affecting people's use of natural places such as woodlands.

The focus of this report section within the wider context of 'access to nature'

As this report is focused on the UK's woods and trees, this section will focus on woodlands, acknowledging they form just part of the overall provision of different natural environments which also contribute to health and wellbeing (Natural England, 2024). Woodlands are in the top three most visited type of natural spaces in England (Natural England, 2024), Northern Ireland (DAERA, 2022) Scotland (Stewart & Eccleston, 2020) and Wales (Natural Resources Wales, 2017) and appear to be an important habitat type when it comes to delivering wellbeing benefits (Goodenough & Waite, 2020; Maes et al., 2021).

What is the economic value of the health benefits provided by UK woodlands?

Benefits to mental health

Forest Research published a research report in 2021 (Saraev et al., 2021) which presented the first (and most recent) attempt to estimate the mental health benefits associated with the UK's woodlands. They used an 'avoided costs' approach to estimate the value of woodland through the reduced prevalence of mental illness brought about through the public visiting woodland.

The values are based on evidence of the reduced incidence of depression and anxiety as a result of regular visits to nature, using data on the number of regular visitors to woodlands, and the prevalence of mental health conditions in the general population, to estimate the number of cases which may be reduced. The avoided costs are based on the average annual costs to society of living with depression and anxiety. Costs include costs of treatment such as GP visits, drug prescriptions, inpatient care and social services, in addition to employment-related costs based on days of work lost to poor mental health.

The annual mental health benefits associated with visits to the UK's woodlands were estimated to be £185 million per year (at 2020 prices) (Saraev et al., 2021).

Forest Research also produced a breakdown of the estimate at the country level, based on population size (rounded to the nearest million). These are presented in Table 1.

Table 1. Estimated annual value associated with visits to woodland in terms of avoided costs of mental illness for the UK, and broken down by country based on population size. Values are based on 2020 prices and rounded to the nearest million. Source: (Saraev et al., 2021).

UK total	England	Northern Ireland	Scotland	Wales
£185 million	£141 million	£6 million	£26 million	£13 million

Benefits to overall health

The Office for National Statistics presents an annual valuation of the overall health benefits from recreation in woodlands (ONS, 2024a). It uses data on the number of people visiting woodland, and the duration of those visits, the health benefits from exercise and exposure to nature, and the cost-savings these provide for the NHS (ONS, 2024b).

The annual valuation of overall health benefits from recreation in woodlands was estimated at £1.149 billion in 2022 (Office for National Statistics, 2024). This is based on people visiting for at least 120 minutes a week, as this has been shown to be associated with health and wellbeing benefits (White et al., 2019).

The Office for National Statistics also produced a breakdown of the annual valuation at the country level, these are presented in Table 2.

Table 2. Annual value of health benefits gained in 2022 from recreation in woodlands (2022 prices). Source: (ONS, 2024c).

UK total	England	Northern Ireland	Scotland	Wales
£1,149 million	£859 million	£28 million	£163 million	£ 99 million

How many people are regularly visiting woodlands?

The number of people regularly visiting woodlands in the UK is tracked through time by two surveys: the Public Opinion of Forestry Survey (Forest Research, 2023), and the People and Nature Surveys (or equivalent) for the four countries of the UK (DAERA, 2022; Natural England, 2024; Natural Resources Wales, 2017; Stewart & Eccleston, 2020). The repeated collection of this data can help show the future impact of any interventions to improve provision of accessible woodland on the population's visiting frequency.

Each survey for the four countries poses the questions on the questionnaire slightly differently, so the percentages show different things and can't be meaningfully compared. The only survey presenting a UK-wide statistic is the UK-wide Public Opinion of Forestry Survey, so this has been selected for presentation here. The findings from this survey cannot be broken down into all countries however because the sample sizes would be too small for Scotland, Wales and Northern Ireland. Separate surveys for the four countries were carried out which give country-specific stats (Forest Research, 2023d, 2023b, 2023a, 2023c), but due to differences in methodology they are not comparable between all countries.

According to the latest UK-wide Public Opinion of Forestry Survey (Forest Research, 2023):

74% of respondents reported they had visited woodland in the last few

years.

34% of people who had visited woodland in the last few years reported having visited woodland several times a month or more during the previous summer.

23% of people who had visited woodland in the last few years reported having visited woodland several times a month or more during the previous winter.

Table 3. Percentage of survey respondents and their reported frequency of woodland visits in the previous summer (April to September), and previous winter (October to March), across the UK. The total sample of respondents is those who indicated in a previous question on the survey that they had visited woodland within the last 'few years'. Therefore, there is a portion of people who never visit woodlands who are excluded from these percentages. 'More than once per month' was not a response option on the survey – percentages of people visiting 'more than once per month' presented here have been calculated by summing the percentage of people responding with 'several times per week' and 'several times per month' to summarise these groups into a single group of frequent woodland visitors, using more than once per month as an arbitrary cut off point at which to consider someone a frequent woodland visitor.
Source: (Forest Research, 2023)

Year of survey	2013	2015	2017	2019	2021	2023
Visited woodland more than once per month in the previous summer	47%	44%	46%	49%	42%	34%
Visited woodland more than once per month in the previous winter	24%	24%	25%	24%	34%	23%

The fluctuations in visiting rates over the last 10 years may be, at least in part, due to marked changes in sample size between surveys. The 2023 survey had a very high sample size compared to previous years and therefore is potentially a more reliable snapshot of the population (Forest Research, 2023). In 2021, there was an exceptionally high number of people (34%) reporting regular visits during the winter of 2020, during which the UK was in various stages of lockdown due to the covid-19 pandemic.

The percentage of people who visited a wood in the last few years reporting visiting woods regularly during winter is consistently lower than it is for the summer (Forest Research, 2023). The most popular activities for woodland visits were walking and dog-walking in both seasons, with a similar proportion of walking and dog-walking in summer and winter (Forest Research, 2023). This finding is interesting as it suggests there are a core of people who incorporate regular woodland walks as part of their life throughout the year, whether that involves a dog or not. The People and Nature Surveys for England, Wales and Northern Ireland also show that fresh air, physical exercise, mental health and wellbeing are cited as reasons for visiting woods throughout the year, even more often than dog-walking is (Natural England, 2024; Natural Resources Wales, 2017), though dog-walking is the most frequently given reason for visiting the outdoors according to the People and Nature Survey for Scotland (Stewart & Eccleston, 2020).

As people tend to travel shorter distances to visit woodlands in the winter (Forest Research, 2023) local access to woods may be of even greater importance at this time of year. Research to link people's Woodland Access Standard status to their frequency and seasonality of woodland visits could help to understand the relative impact that proximity to woodland has on influencing people's behaviour when it comes to maintaining a lifestyle which includes regular woodland access year-round, compared to other factors. The factors influencing the frequency of woodland visits are of interest because it is reasonable to suspect that the wellbeing benefits of woodland access, just like other healthy habits, would be maximised via a pattern of regular and sustained engagement, compared with infrequent exposure to woodland limited to half the year. However, it should be noted that those who do not visit woodlands may still receive wellbeing benefits from other natural environments (DAERA, 2022; Natural England, 2024; Stewart & Eccleston, 2020), as woodlands form just part of a wider 'infrastructure' of green and blue space.

What prevents more people from visiting woodlands?

Results presented above from Forest Research's latest Public Opinion of Forestry survey for the UK (Forest Research, 2023) show that over a quarter of the population in the UK have not visited a woodland 'in the last few years'. Therefore, there is potential for expanding woodland wellbeing benefits to unreached parts of the population.

There are many reasons why people may not visit woodlands regularly. A pair of recent literature reviews (Pearson et al., 2023; Gardner 2023) summarise evidence on the barriers which can prevent or deter people from visiting woodlands. The key findings are presented in Table 4.

Table 4. Summary of the findings of an evidence review by Forest Research on the barriers to visiting woodlands experienced by diverse publics. Source: (Pearson et al., 2023)

Barrier	Details	References
Distance from accessible woodland.	The further someone's residence from woodlands, the less often they visit. Cost of transport and the time taken to travel can pose a barrier to visiting green spaces, especially for people dependent on public transport. Lack of public transport and parking costs also pose barriers.	(Dallimer et al., 2014; Žlender & Ward Thompson, 2017; (Parks for London, 2022:36; O'Brien & Forster, 2020).
Quality of woodlands	Good directional signs and information boards and freedom from rubbish make the greatest difference to people accessing local woodlands. Preferences for facilities/infrastructure are distinct to different user-groups. Quality of woodland structure and nature are also important, with a preference for mixed forest stands over monocultures, and for woodlands which are neither under nor 'over' managed – allowing woodlands to feel both safe and natural.	(Ward Thompson et al., 2005; Dallimer at el., 2014; Hall et al., 2022).

Barrier	Details	References
Lower income/ socioeconomic status	Higher income is associated with individuals visiting woodland more often, other factors constant. Adults from lower socio-economic status in deprived areas with less greenspace, constitute infrequent users of woodland.	(Boyd et al., 2018)
Lack of interest	Childhood experience of accessing woodland is associated with use of woodland later in life. Those who access woodlands as children tend to be from higher socioeconomic backgrounds and thus the legacy of lack of childhood access is a barrier especially for those from lower economic backgrounds and areas. 'Lack of interest' in visiting woodlands has been found to be more highly self-reported among ethnic minority groups.	(Thompson et al., 2008; Winter et al., 2019)
Poor health	Those in poor health are less likely to use greenspace, including woodland.	(Boyd et al., 2018; Public Health Scotland, 2022; Thalén et al., 2022)
Physical disability	Physical disability was one of the top eight reasons self-selected by individuals in the People and Nature survey for not accessing greenspace, including woodland.	(Boyd et al., 2018)
Age	Teenagers access woodlands less than adults do.	(Hegetschweiler et al., 2022)
Lack of time	Time availability is a common barrier to woodland access. Factors such as long work hours and caring responsibilities limit access to woodland.	(Boyd et al., 2018; Winter et al., 2019; O'Brien, 2019)
Gender	Women are less likely to access woodlands than men in general. Young women particularly cite fear around becoming victims of assault or attack due to lack of visibility in woodlands as a barrier to access.	(Boyd et al., 2018; Milligan & Bingley, 2007; Sonti et al., 2020)
Ethnicity	People in minority ethnic groups are under-represented in woodland environments and can feel unwelcome or excluded. Lack of inclusive imagery can contribute to this. Insufficient facilities for social gathering can pose a barrier, as can unfamiliarity with the area. Prior experiences of discrimination, and the fear of discrimination, including violence, can also be barriers.	(Ferguson et al., 2018; Winter et al., 2019; Armstrong & Greene, 2022; Burgess 1996; Edwards, Larson, & Burdsey, 2022)

Among the many interacting factors which combine to explain why some people do not access woodlands (see Table 4), low socioeconomic status appears to be a common thread which interacts with many of them. For example, more deprived neighbourhoods generally have less local access to greenspace, including woodland (Friends of the Earth, 2020), meaning distance is more of an issue. The availability and costs associated with transport then become more relevant. Low socioeconomic status increases one's risk of poor health (Lewer et al., 2020), which can then pose a barrier to visiting woodlands. Being less likely to have experienced visiting woodlands as a child, coming from a lower socioeconomic background may then lead to an intergenerational cycle of disinterest in visiting woodlands through lack of experience of the wellbeing benefits, and a sense they are not safe, familiar, welcoming, or normal places to visit.

Some other types of green/bluespace may be more suitable for meeting the nature-access needs of particular groups or individuals based on their requirements or preferences, but efforts should still be made to make woodlands as inclusive and accessible as reasonably possible.

How many people have access to a woodland near where they live?

The presence of nearby nature is fundamental

To receive the greatest benefits from nature, people need to experience nature regularly. As shown in Table 4, the time taken to reach greenspace areas, and costs and availability of transport, are some of the most important determinants of whether people access greenspace or not, especially in more deprived areas (Pearson et al., 2023). Therefore, a foundational requirement to enable regular nature experience is the availability of natural environments which are open to public access, such as publicly accessible woodlands, within a reasonable distance of people's homes. The distance considered 'reasonable' by the Woodland Trust is set out in the Woodland Access Standard, see below. Understanding the state of people's proximity to accessible woodlands is critical for guiding interventions to improve access.

What should we aim for in terms of proximity to accessible woodlands from people's homes?

The Woodland Trust has advocated for reasonable proximity to woodlands for everyone in the UK, defined by the **Woodland Access Standard**, which aspires that:

- no person should live more than 500m from at least one area of accessible woodland of no less than 2ha in size; and
- there should also be at least one area of accessible woodland of no less than 20ha in size, within 4km (8km round trip) of people's homes.
- In this context, the word 'accessible' is used to mean woodland which the public are permitted to access, rather than relating to accessibility in the context of mobility.

There are two important aspects to determining whether people meet the Woodland Access Standards – where accessible woods are, and how many people live close enough to them.

United Kingdom

The state of accessible woodland mapping across the whole of the UK

The best available UK-wide mapping data for accessible woodland across the UK is the Woodland Trust's Woods for People dataset, reported in the State of the UK's Woods and Trees report in 2021. The dataset has not been updated in its entirety since the previous analysis.

This data came from the Woods for People project, begun by the Woodland Trust in 2002 in partnership with the Forestry Commission. The aim of that project was to produce a comprehensive inventory of accessible woodland across the UK. The dataset included mostly accessible woodland in public (NGO and Local authority) ownership but also included some privately owned accessible woodland sites. Inclusion of landowners' woodlands within the dataset is entirely voluntary. Since the end of the initial project to create the dataset, the Woodland Trust has attempted to maintain it by keeping in touch with all the landowners. However, this has proved challenging, and it is unclear how comprehensive the Woods for People dataset remains two decades since its creation.

The Woods for People data defined accessible woodlands as "any woodland that is permissively accessible to the general public for recreational purposes" This includes sites with unrestricted open access and restricted, but permissive, access e.g. fee-payable, fixed-hours access. The Woods for People definition does not include woods which are only accessible due to the presence of a public right of way running through them, as opposed to full public access. This is because when Woods for People first began, digital data on rights of way was not easily accessible and research had shown that walkers often experience difficulties of access on rights of way. It is likely this definition led to underestimation of the availability of woodlands which the public are able to experience and benefit from. While the Scottish Outdoor Access Code means theoretically all woodland in Scotland is accessible to the public, the Woods for People dataset for Scotland only included woodlands which had been voluntarily put forward for inclusion, as the aim was to record those woodlands where public access is welcomed rather than tolerated.

What we know about how many people meet the Woodland Access Standards across the UK

The whole dataset has not been updated since the previous analysis presented in the previous State of the UK's Woods and Trees report (Reid et al., 2021), so no new UK-wide analysis has been undertaken and there are no new results to report here.

England

The state of accessible woodland mapping in England

For England only, a new, more comprehensive dataset on the location of accessible woodlands was developed in by the Forestry Commission in 2023/24. The new dataset, which the commission has named Woods for All, includes the Woodland Trust's Woods for People data for England (which the Forestry Commission provided funding and human resource support to update in 2023) and builds on this by incorporating additional data on:

- Where public rights of way pass through non-accessible woodlands.
- Where access is supported through grant schemes such as the England Woodland Creation Offer and Countryside Stewardship.
- Additional accessible woodlands identified within other accessible greenspace datasets.

The Woods for All dataset provides a much more comprehensive inventory of the current extent and location of woodland which people can visit, compared to the Woods for People data alone. By including the distance of public rights of way through otherwise non-accessible woodlands, the Woods for All dataset drastically increases the amount of woodland access recorded.

As the natural environment and access to the countryside is a devolved issue, the work described above to update the data for England does not include any updates to the Woods for People data within the other three nations of the UK.

What we know about how many people meet the Woodland Access Standards in England

For England, DEFRA's Environmental Improvement Plan published in 2023 includes the commitment to ensure that anyone can reach a green or blue space within 15 minutes of their front door, and states that woodlands will play a key part in delivering this.

DERFA's Woodland Access Implementation Plan 2024 acknowledges that improved baseline data is required to understand existing woodland access provision, so that this can be used to target increased woodland access and track progress towards achieving the commitment for England. As such, the Forestry Commission has been working to develop the existing baseline datasets and DEFRA's Nature For Climate Fund has funded new research by Forest Research which includes testing new ways of measuring how many people meet the Woodland Access Standards in England.

This new analysis by Forest Research is currently underway and yet to be published at the time of writing this report, so no findings are available for reporting here. It will include information on how the level of local woodland access varies across different areas in England, and whether woodland access is equitable across socioeconomic groups in England by analysing the relationship between the socioeconomic status of areas and how many people meet the Woodland Access Standards.

This is important because previous work focusing on greenspaces across Europe has shown the availability of local green space tends to be unequally distributed across society (Schüle et al., 2019), with those who could benefit the most, having the least availability. This is also the case in the UK (Friends of the Earth, 2020; Ngan et al., 2024) A key question for the state of woodland access therefore is to understand whether woodland access is equitable across socioeconomic groups within the UK.

Scotland

The state of accessible woodland mapping in Scotland

Although Scotland's Outdoor Access Code means technically all woodlands in Scotland are publicly accessible, the Woods for People data included only woodlands where the landowner had expressed they welcome visits from the public, rather than including all woodlands. This approach means the same challenges of keeping the accessible woodland data updated exist for Scotland as for the other countries. The data has not been fully updated since the version presented in the last State of the UK's Woods and Trees report in 2021. The National Forest Inventory data which shows the location of all woodlands could be used as a dataset on accessible woodlands for Scotland due to the Scottish Outdoor Access code.

What we know about how many people meet the Woodland Access Standards in Scotland

It would be possible to analyse the proportion of people meeting the Woodland Access Standards in Scotland using data from the National Forest Inventory on the location of all woodlands and their proximity to people's homes. However, this has not been done, as the Woodland Trust did not have capacity in-house to undertake any analysis of the proximity of people to woodland for this report.

Northern Ireland

The state of accessible woodland mapping in Northern Ireland

Outscape, a not-for-profit organisation, created a map called GreenspaceNI, which shows the location of greenspace within Northern Ireland, and what proportion of that greenspace is woodland. The Greenspace NI Map was launched in June 2023 and will be updated annually. The map is available here Greenspace NI Map Viewer.

The map was created with funding from The Department of Agriculture, Environment and Rural Affairs (DAERA) and the Department for Infrastructure (DFI) on behalf of the Cross-Governmental Strategic Outdoor Recreation Group (SORG)

Details of the data used to create this map can be found here:

GreenspaceNI Map Product Overview

GreenspaceNI Map – Metadata.

What we know about how many people meet the Woodland Access Standards in Northern Ireland

The Greenspace NI data has been used to calculate how many households are within 400m of a greenspace and trails (Outscape, 2024) but no analysis specifically focused on proximity of households to woodlands has been undertaken using this dataset, however it should be possible, as woodland is a category of greenspace within the dataset.

Wales

What we know about how many people meet the Woodland Access Standards in Wales

No analysis of how many people have local woodland access in Wales has been undertaken since the previous State of the UK's Woods and Trees report in 2021.

What impact on the state of woodland access has policy-change made since the last report?

In England a new publicly funded grant scheme called the England Woodland

Creation Offer (EWCO) came into effect in 2021. The scheme aims to incentivise land managers to create woodland to deliver various public benefits, one of which is recreational access. Landowners can opt in to receive an additional payment to support public recreational access to the new woodland they are creating with their EWCO grant. The England Woodland Creation Offer came into effect in June 2021. In the period between then and September 2023 when the last official statistics were released, only 8% of land managers receiving a EWCO grant took up the optional recreational access contribution representing 270ha of confirmed accessible woodland creation since EWCO launched in June 2021 (DEFRA, 2024).

In England, DEFRA's Woodland Access Implementation Plan (WAIP) was published in November 2023 and various actions are underway to improve the quantity, quality and permanency of access to woodlands. For example, Forestry England has announced that it will buy new areas of land to create the Coronation Woods, providing new opportunities for woodland open access. The WAIP is delivering a range of research to inform how best woodland access can be delivered.

Also in England, DEFRA is funding a new Forest for the Nation, to build on the success of the National Forest in central England, which aims to enhance public access to woodlands.

In Wales, as part of the National Forest for Wales programme, the Tiny Forests/ Coetiroedd Bach Grant will create 100 'tiny forests' between April 2023 and the end of March 2025. All these new woodlands will include public access as a condition of the grant.

In Northern Ireland, DAERA recently published its Environmental Improvement Plan for Northern Ireland which states a target of an "annual increase in % of households that have publicly accessible quality natural space >2ha within 400m and at least one site >20ha in size within 2km, with the target of achieving 84% of households meeting this standard by 2050, and 90% of households visiting the natural outdoors at least once a week" (DAERA, 2024). This commitment drove the development of the GreenspaceNI data so that progress towards this target can be measured going forward.

What prevents more land managers from providing public access to their woodlands?

A large proportion of woodlands in the UK are owned by private landowners. It is important to understand the needs and attitudes, concerns and barriers which may be preventing landowners from permitting access to their woodlands, so that policies can be designed to effectively support landowners to increase public access to existing woods.

A review of the evidence for England by Forest Research (McConnachie & Gardner, 2024) summarised the evidence around land managers' attitudes to public access to woodlands. This is synthesised in table 5. The review highlighted that much of the evidence on land manager attitudes to public access is out of date – mainly over 20 years old. There is a lack of evidence evaluating the impact of existing interventions to support land managers to increase public access provision. To address the need for more up-to-date information on this topic, Forest Research and the Sylva foundation are currently surveying woodland owners, woodland managers and forestry professionals across Britain to gain a better understanding of their current attitudes to public access to woodlands, and current barriers faced.

Table 5. Barriers and issues concerning woodland managers with regards to public access to woodlands. Summarised from evidence review by McConnachie & Gardner (2024).

Economic implications	• Costs of installing and maintaining public access infrastructure and public liability insurance premiums (Christie-Miller, 2000; Church et al., 2005; MacKay & Prager, 2021; Thompson, 2021; Urquhart et al., 2010). • Cost of managing the impacts of litter, vandalism, and dogs – mostly in highly visited sites (Crabtree, Chalmers, and Appleton, 1994). • Loss of income from public access impacting income generating uses of the land such as timber operations and shooting. • Public access provision may negatively affect land values, with exceptions for cases when it raises business opportunities (Addland, 2023; Buckley et al., 2008).
Public liability	• Concern and uncertainty around a land manager's liability for injuries members of the public may sustain when accessing their land (Christie-Miller, 2000; Sime et al., 1993; Urquhart et al., 2010; Probert 2005; Thompson, 2021).
Social issues	• Theft and vandalism (Christie-Miller, 2000; Church & Ravenscroft, 2008; Costley, 2001; Sime et al., 1993). • Damage to property including crops, walls, fences and gates (Warren, 2002) or to farm animals, crops and machinery (Costley, 2001). • Litter (Christie-Miller, 2000; Church & Ravenscroft, 2008; Costley, 2001; Thompson, 2021; Warren, 2002; Williamson, 2001). • Dogs – fouling, disturbing livestock and wildlife (Christie-Miller, 2000, p. 208; Costley, 2001; Nicholls et al., 2013, p. 37; Thompson, 2021, p. 30; Williamson, 2001). • Visitors straying from designated paths and leaving gates open (Church & Ravenscroft, 2008; Nicholls et al., 2013; Thompson, 2021) • Illegal vehicular access. • Disturbance of game birds (Church & Ravenscroft, 2008).
Bureaucracy	• Cost, time and effort associated with grant applications and management (Dandy, 2012; Lawrence et al., 2010; Molteno et al., 2012; Urquhart et al., 2010).
Permanency of commitment	• Concerns around public access being a permanent land use change (Molteno et al., 2012, p. 48; Thompson, 2021, p. 70).

How does the quality of woodlands influence people's wellbeing?

Beyond simply being able to access woodland, the quality of woodland can affect people's wellbeing experience. This can include the quality of infrastructure, for example paths, car parks, toilets and benches, especially for some participants, particularly the elderly, disabled and those suffering from long-term health problems (O'Brien et al., 2012). If green spaces feel or appear neglected, this is important in whether people choose to access the space or not (Pearson et al., 2023). Freedom from rubbish, good quality paths and provision of waymarkers and information boards are important to people visiting woodlands (Pearson et al., 2023).

A review of greenspace characteristics and their influence on mental health found that distinguishing which types or characteristics of green space are most beneficial for mental health is difficult to answer with the current evidence base (Beute et al., 2023). However, one pattern which did emerge was that vegetation structure can be influential: a higher density of vegetation can have a negative effect on mental health as it can reduce visibility and feelings of safety (Beute et al., 2023). Other studies also present findings showing a general preference for a vegetation structure which is varied or medium to low density, striking a balance between 'managed' and 'wild' (O'Brien et al., 2012; Chiang et al., 2017; Liu et al., 2022). Participants in a study in England favoured variety and complexity in the woodland environment including some open spaces that gave them views across the landscape (O'Brien et al., 2012). A wider study across 12 European countries including Scotland, also found a strong preference across all surveyed countries for more structurally complex forests; older stands with diverse tree species (Giergiczny et al., 2024).

Biodiversity and wellbeing

Through a combination of improved provision and advertising of local accessible woodlands, interventions to address social/cultural barriers to access, and improved woodland infrastructure/management, we may increase the number of visits to woodlands and get people feeling safe and comfortable while there. That would be significant progress, however that may only be part of the solution. The ecological quality of greenspaces can also be important for the wellbeing benefits they provide for people when they visit (Bell et al., 2018; Fuller et al., 2007; Houlden et al., 2021; Knight et al., 2022; Wyles et al., 2019). But what exactly does a high-quality woodland for people's wellbeing look like from an ecological perspective?

Research investigating the role biodiversity plays in delivering wellbeing is relatively sparse. Studies have mostly focused on the effect of greenspace area, proximity and accessibility on wellbeing, without considering biodiversity value of greenspaces (Hooyberg et al., 2020; Van Den Eeden et al., 2022). Yet the ecological condition of woodlands can differ dramatically, affecting the biodiversity present. It is challenging to study how biodiversity influences wellbeing because the concepts of biodiversity and wellbeing are both complex and can be defined in many ways making it difficult to draw broad conclusions about the relationship between biodiversity and wellbeing (Hedin et al., 2022).

Findings from the few studies that have examined objective metrics of biodiversity, such as species richness and abundance, show inconsistent outcomes. For example, Dallimer et al., (2012) found variable relationships between people's wellbeing and the actual species richness of birds, butterflies and plants in greenspaces. However, the same study showed a positive relationship between people's wellbeing and their *perception* of species richness in the greenspace.

This suggests that people want to interact with biodiversity, but don't relate to it in terms of the objective ecological metrics commonly used to study greenspace quality. Recognising this, researchers have drawn on social science techniques to explore this conundrum and shown that the sensory traits of biodiversity (for example, colours, sounds, smells, textures) are key for how people relate to and derive wellbeing from woodlands (Bentley et al., 2023; Fisher et al., 2023). The influences of culture (e.g. literature, films, gaming characters, comic superheroes), community (e.g. family, friends) and personal experiences of previous encounters are central to people's preferences and perspectives on biodiversity (Austen et al., 2021, 2023; Fish et al., 2024). Therefore, particular aspects of biodiversity which appeal to the senses, can have an influence on the wellbeing experience of visiting woodland. For example, broadleaf trees with the changes in colours and textures they give through the seasons, bird song, the smells of damp vegetation, decaying wood and wild garlic (Austen et al., 2021, 2023; Fisher et al., 2023).

To improve our understanding of how biodiversity contributes to wellbeing, researchers have recently developed a new, reliable and validated self-reported wellbeing scale for investigating the biodiversity-health/wellbeing relationship (Irvine et al., 2023; Jones et al. 2024). The scale is called BIO-WELL BIO-WELL: the biodiversity and human wellbeing scale and it enables researchers to quantify the wellbeing effects people experience from biodiversity, to enable policy makers and practitioners to make evidence-based decisions on how to improve wellbeing through nature.

As part of a Woodland Trust-funded research project (Fisher et al., 2024, *preprint;* Fisher, 2025a, *manuscript in preparation*) over 5,000 people representing a diverse cross-section of the public from across England, Wales, Scotland and Northern Ireland were surveyed online using BIO-WELL (a psychometric scale designed to quantify people's wellbeing responses to biodiversity). People were asked to imagine themselves in a nearby forest at the current time of year and to think about the living things, including the plants, fungi and animals, then asked questions related to various aspects of wellbeing. Overall, people reported experiencing positive wellbeing in response to biodiversity within a woodland local to them, with an average BIO-WELL score of 61.7 out of 100 (where values above 50 represent positive wellbeing in response to biodiversity).

90% of participants indicated positive wellbeing responses in response to the biodiversity in their local woodlands (Fisher et al., 2024, *preprint*).

How does the biodiversity-wellbeing quality of woodlands vary across the UK?

The support biodiversity provides to the wellbeing of the public can be considered a vital ecosystem service (Bratman et al., 2019b). In functional ecology, species traits which deliver ecosystem services are called 'effect traits'. For example, the length of an insect's proboscis may be an important effect trait for its role in providing the service of pollination. Effect traits are the elements of biodiversity which provide the mechanism by which ecosystems deliver ecosystem services. In the same way, species' traits which induce feelings of wellbeing in people can be considered 'effect traits' – providing the mechanism by which the wellbeing benefits from ecosystems are delivered (Fisher et al., 2023). Effect traits can be thought of as the elements of biodiversity which trigger wellbeing-boosting sensory experiences, such as smells, colours, textures, sounds, shapes and wildlife behaviours (Fisher et al., 2023).

The overwhelming majority of woodland species' effect traits deliver positive wellbeing, particularly spiritual wellbeing, but also physical, emotional, cognitive, social and overall wellbeing. Some species can be thought of as 'keystone species' for their services to wellbeing, as they exhibit a disproportionate number of unique positive effect traits (sounds, colours, behaviours, textures and smells), particularly silver birch *Betula pendula,* horse chestnut *Aesculus hippocastanum* and pedunculate oak *Quercus robur*. However, effect traits of other species can induce negative wellbeing (such as ticks *Ixodes ricinus*). Each additional species within a woodland brings with it additional effect traits, the majority of which are positive, showing that diversity in woodland ecosystems is beneficial for the wellbeing of people visiting them (Fisher et al., 2023). As with any ecosystem service, degraded woodlands are likely to be sub-optimal in their provision of wellbeing, due to missing effect traits - though which traits are most important and whether thresholds or tipping points exist remains unknown.

Considering what we know about the sub-optimal ecological condition of much of our woodland in the UK, and the declines in the special wildlife which should inhabit them (elsewhere in this report) many UK woodlands are unlikely to be delivering the level of wellbeing benefits they could if their ecological communities were intact. Where woodlands are supporting a greater diversity of species due to better ecological condition, they are likely to also provide greater wellbeing experiences for those who visit them. It is not known whether the woodland biodiversity which can underpin wellbeing experiences is distributed equally across society.

As part of a project funded by the Woodland Trust, a team of researchers at the University of Kent mapped the distribution and cumulative richness of effect traits that deliver physical, cognitive, emotional, social and spiritual wellbeing across woodlands in the UK, by overlaying distribution models for the species which provide the effect traits. The map reveals the spatial distribution of woodland quality in terms of modelled capacity to provide rich wellbeing experiences. This was combined with data on neighbourhood level deprivation to assess inequality in this predicted woodland-wellbeing quality at a local level.

These maps by the University of Kent (Fisher et al., 2024, *preprint;* Fisher, 2025a, *manuscript in preparation*) show the cumulative modelled distribution of woodland species (including birds, butterflies, fungi, mammals and plants) known to possess wellbeing effect traits (Austen et al., 2021). As such, the maps indicate the richness of these wellbeing effect traits across England and Wales, Scotland and Northern Ireland. See figure 1.

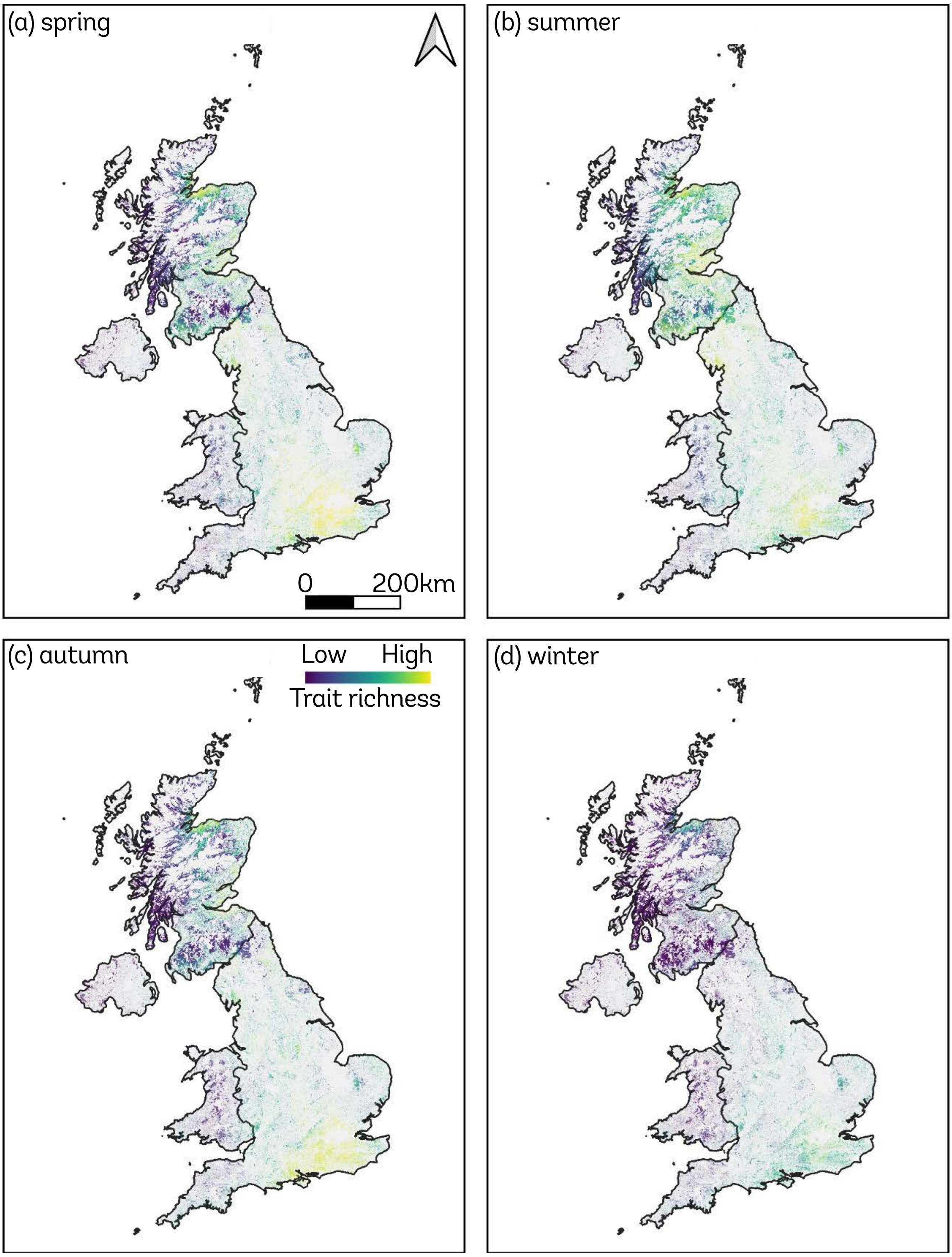

Figure 1. Cumulative richness of seasonal species' effect traits with the potential to elicit positive human wellbeing responses, across forests in the United Kingdom. NB: Different datasets were used to generate the species distribution models in Northern Ireland and Scotland, compared to England and Wales, due to differences in data availability in the devolved nations. The data is scaled from 0 (low) to 100 (high) to enable approximate comparisons (Fisher et al., 2024, preprint; Fisher, 2025a, manuscript in preparation).

Species distribution maps were generated by taking the known occurrence locations of species and applying statistical models to identify relationships between these locations and key environmental variables (temperature, rainfall, soil and elevation). These models were then used to predict the likelihood of species presence across England and Wales, producing distribution maps. These individual species maps were combined to produce a single species richness map and then linked to the wellbeing traits these species possess.

Note that the effect trait richness maps are based on species distribution models. These models predict the presence or absence of a species based on the statistical relationship among locations where a species is known to be present (though biological records), and environmental variables, such as temperature, rainfall, soil and elevation. The trait richness maps therefore indicate areas with potentially suitable conditions for the species. The trait richness value predicted for an individual woodland is reflective of the broader landscape scale, rather than the unique properties of that particular woodland. The modelling does not take into account the ecological condition of individual woodlands. Field surveys to confirm the presence or absence of the species at individual woodlands would be required to confirm the accuracy of the model prediction at that scale. Also, interpreting the modelled cumulative effect trait richness as an indicator of wellbeing potential of a woodland makes the assumption that all effect traits are of equal weighting or importance for wellbeing, and that the wellbeing value of woods is linear in its association with cumulative trait richness. It is not known whether some effect traits are more important than others or whether thresholds or tipping points exist in the relationship between effect trait richness and the wellbeing benefits people receive. Further research building on this work would be needed to understand these details.

It is important to clarify that, as this research has a woodland focus, the wellbeing-effect traits mapped in this work relate to species associated with woodlands. Areas with poor woodland effect trait richness may have other highly valued habitats supporting different species which also bring their own unique wellbeing traits such as coastal, mountain or moorland environments. Whilst these habitats and species are also important for wellbeing, they are not represented in this spatial analysis of woodland wellbeing traits. Further work to map wellbeing effect traits associated with other habitat types would help to set these findings in the context of benefits people receive from the total variety of natural spaces available, to identify the highest priority areas for nature restoration once these other habitats are also taken into account. For example, people in the South West may not have high effect trait richness from woodland species but benefit from effect-traits supported by coastal species experienced on clifftop walks or visits to the beach.

The trait maps show different effect trait richness values across the UK, suggesting the potential for woodlands in different regions to deliver wellbeing may be unequal (Fisher et al., 2024, *preprint*; Fisher, 2025a, *manuscript in preparation*). The high concentration of wellbeing effect traits in the South East of England and some areas of Scotland reflects that many of the woodland species identified as beneficial for their effect on wellbeing (Fisher et al., 2023) are predicted by the species distribution models to occur within those regions. However, it should be noted that distribution models were not generated for all the species identified as relevant for wellbeing. This was due to lack of data availability for some species. Therefore, it is possible that the location of high concentrations of wellbeing effect traits could be an artefact of which species had available data.

This analysis does not provide any direct evidence as to why the South

East of England and some parts of Scotland would be particularly rich in the woodland-associated biodiversity underlying the wellbeing trait-richness map. The hotspot in the South East could reflect that most species tend to be on the northwestern edge of their European range in the UK, so more species are present in the South East. It could also reflect that the South East landscape is characterised by high broadleaved woodland cover (Forestry Commission, 2002b). For example, this landscape features the High Weald, which is the most wooded Area of Outstanding Natural Beauty (AONB) in the UK, with a high proportion of ancient woodland (High Weald National Landscape Partnership, 2024), see https://highweald.org/about-the-landscape/woodland/ . We know from analysis presented in State of the UK's Woods and Trees 2021 (Reid et al., 2021) that the extent of broadleaf woodland cover is closely correlated with woodland plant species richness for any given area, whereas no similar correlation exists with coniferous woodlands. Since each woodland plant species may bring multiple wellbeing traits in the form of colours, textures, sounds, smells and behaviours, this increased richness in woodland plants in the South East of England is likely an important factor contributing to the high wellbeing trait richness predicted for this region.

As an example, Surrey is a county in the South East of England which the effect trait richness map showed as having very high effect trait richness. 21% of Surrey is covered by the priority habitat types 'mixed deciduous woodland' and 'beech and yew woodland', and 3% of its land covered by coniferous woodland (Waite, 2017). Wood pasture and parkland, (with its ancient and veteran trees) is also particularly well-represented in the county (Waite, 2017).

In contrast, areas where the landscape is characterised by low overall woodland cover, such as the East Midlands (Forestry Commission, 2002a) appear to exhibit a pattern of low wellbeing effect trait richness at a regional scale. For example, Lincolnshire in the East Midlands shows very low effect trait richness on the map. This county's landscape is dominated by arable agriculture with very low overall woodland cover. It is one of the least-wooded counties in the UK. The total area of ancient woodlands which remain are very small and isolated, and many have been historically degraded by re-planting with non-native conifers. The area of ancient semi-natural woodland is around 1% (Greater Lincolnshire Nature Partnership, 2024). In areas such as these, the poor extent and condition of woodland in the landscape could partly explain the loss of the richness of woodland-associated biodiversity underlying the effect trait map. However, it should be noted this discussion poses a purely speculative hypothesis based on the observed characteristics of these counties as case study examples, and detailed research would be required to understand the drivers behind the spatial pattern of trait richness with any confidence.

Understanding that species effect traits play a role in people's wellbeing responses to woodland (Fisher et al., 2023) and knowing the current state of woodland nature across much of the UK (Reid et al., 2021) suggests our landscapes may be of diminished capacity for delivering truly inspiring and uplifting woodland experiences. The degraded state of the extent and ecological condition of woodland in some landscapes could have eroded the diversity of the species that make the woodland experience powerful for people's wellbeing.

As the effect trait richness map is a novel way of visualising a certain aspect of biodiversity, it is not clear what is driving the pattern of effect trait richness distribution and further investigation is needed to clarify this. As trait richness is linked to species richness, and varies at a landscape scale, it would appear reasonable to suggest that improving landscape scale woodland species richness through nature recovery interventions focused on improving the extent and condition of wooded habitats in areas where they are most degraded may be a way of increasing woodland effect trait richness in areas where it is low. The inequality in effect trait richness between different areas presents a strong rationale for the need to recover nature at a landscape scale in order to improve the opportunities for people to experience the potential wellbeing benefits of engaging with rich woodland biodiversity. This should be seen as part of a holistic effort to recover landscape scale biodiversity overall through restoring a rich mosaic of a variety of habitats, all of which will provide their own effect traits.

Managing public access to woodlands alongside biodiversity conservation presents challenges and trade-offs, and public access can conflict with nature conservation aims at the site level (Dertien et al., 2021). However, understanding that public benefits from woodlands may be, in part, contingent on the presence of thriving woodland biodiversity offers an alternative perspective and could provide motivation for further policy support for significant landscape scale nature recovery, which could lead to overall positive outcomes for nature. Nevertheless, any increase in public access should consider the fragility of woodland biodiversity and well-planned management of public access is essential to prevent excessive disturbance to wildlife or degradation of habitats.

Do broadleaved or coniferous woodlands have the greatest potential to deliver wellbeing experiences?

A review of the published literature revealed little evidence or discussion of whether broadleaved woodlands or coniferous woodlands have the greatest benefits for wellbeing, especially in the UK context. It is likely that this is strongly influenced by the preference of the person experiencing the woodland. For example, a focus group study (MacNaghten and Urry, 2001) found attitudes to woodlands varied among different social groups. Participant responses from a focus group study in England suggested broadleaved woodlands were generally preferred to conifer plantations (O'Brien, 2004). Whereas another study (O'Brien et al., 2012) found participants held a general sense of appreciation for trees and wooded landscapes (both broadleaved, coniferous and mixed). A study in China found that both coniferous and broadleaved woodlands had a beneficial effect for reducing stress compared with a built environment, but amongst the multiple indicators of stress reduction measured, there was no clearly favoured woodland type (Yao et al., 2024).

The spatial analysis undertaken as part of the recent Woodland Trust-funded research project (Fisher et al., 2024, *preprint*; Fisher, 2025a, *manuscript in preparation*) investigated this question by testing for difference in the mean richness of wellbeing effect traits (see above section for a definition of wellbeing effect traits) for woodlands classified as 'coniferous', 'broadleaved', or 'mixed' on the National Forest Inventory. According to this analysis, broadleaved woodlands were significantly richer in wellbeing traits than coniferous woodlands or mixed woodlands throughout most of the year with the exception of summer when there was no difference, see Figure 2.

This suggests broadleaved woodlands provide suitable habitat for many of the woodland species identified for their positive wellbeing traits outside of summer. Common broadleaved trees such as birch and oak are themselves associated with an especially high number of positive wellbeing effect traits as identified by Fisher et al., (2023).

However, despite broadleaf forests having a greater variety of species with wellbeing-boosting sensory potential, when people were asked to complete a BIO-WELL questionnaire thinking about their local woodland, they perceived a similar strength of positive wellbeing response, regardless of the type of woodland (Fisher et al., 2024, *preprint;* Fisher, 2025a, *manuscript in preparation).*

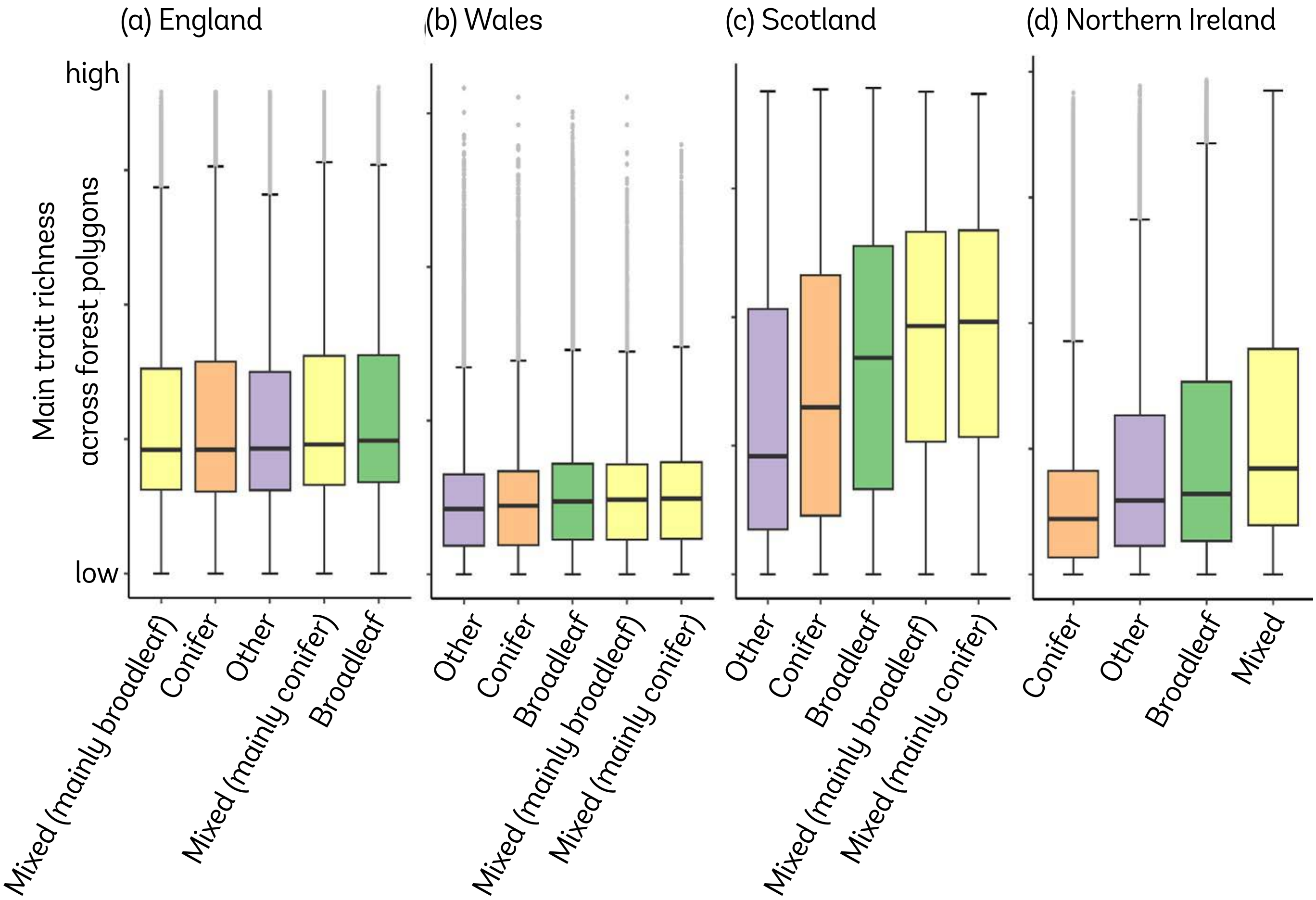

Figure 2 Mean species' effect trait richness for positive human wellbeing, for forest categories across the United Kingdom, averaged across all forest polygons of each category in (a) England (b) Wales, (c) Scotland, and (d) Northern Ireland. Forest categories and colours (yellow = mixed woodland types, orange = conifer, purple = other, green = broadleaf) are derived from the National Forest Inventory and the Northern Ireland Woodland Basemap, respectively. Boxplots depict the median, interquartile range, minimum and maximum of the data, while grey lines represent outliers. NB: Different datasets were used to generate the species distribution models in Northern Ireland and Scotland, compared to England and Wales, due to differences in data availability in the devolved nations (Fisher et al., 2024, preprint; Fisher, 2025a, manuscript in preparation).

Is the biodiversity-wellbeing quality of woodlands in an area linked to the area's socioeconomic status?

Across England, Wales, Scotland and Northern Ireland there is an association between effect trait richness and the level of deprivation of the neighbourhood in which the woodland is located. Woodlands richest in effect traits were located in the least deprived neighbourhoods, and woodlands with fewest effect traits were located in the most deprived areas. This pattern is maintained across all four seasons.

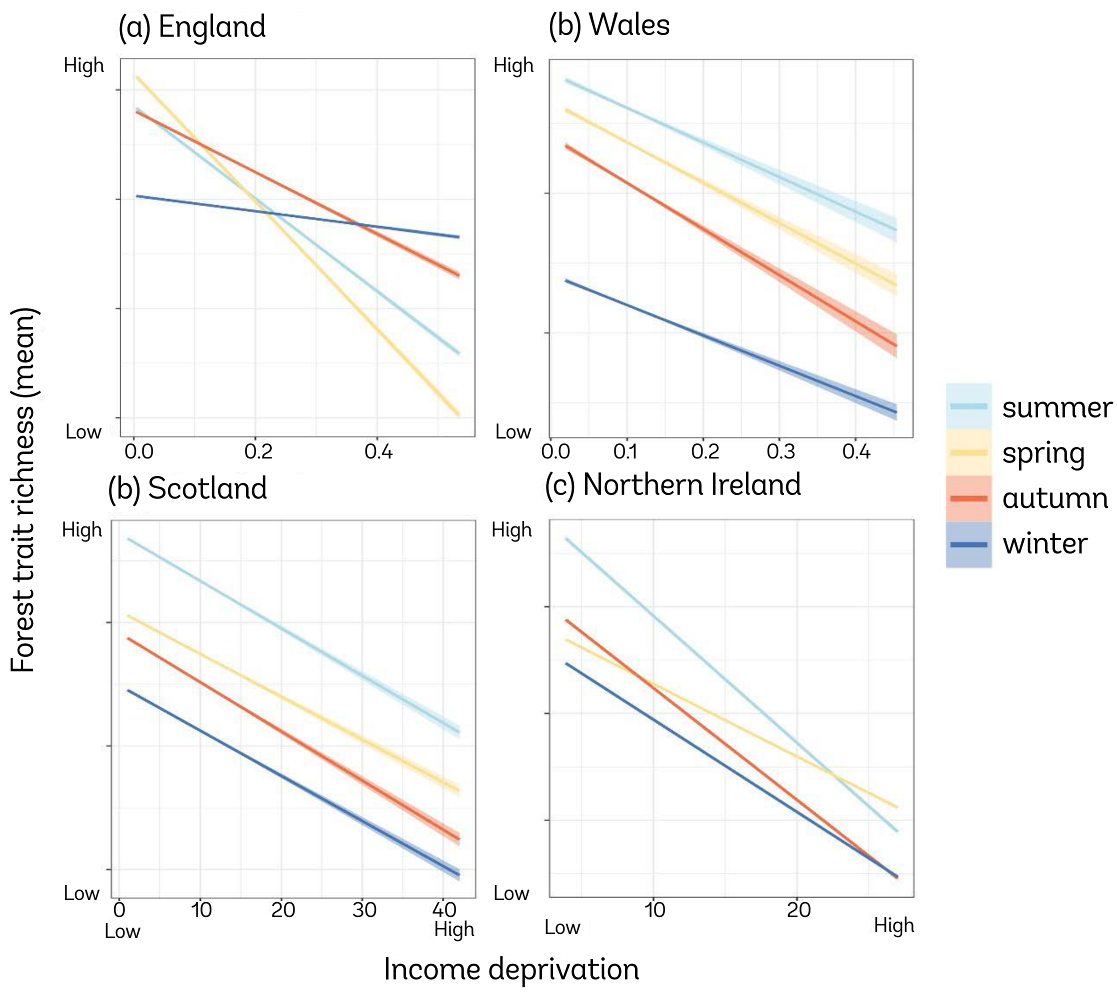

Figure 3 Line graph depicting the linear modelled relationship between species' effect trait richness for positive human wellbeing, and income-related deprivation across the four devolved nations of the United Kingdom, across the four seasons (summer = light blue, spring = yellow, autumn = orange, winter = dark blue). Shading around the lines indicates 95% confidence intervals. Deprivation values of the LSOA are extracted at the midpoint of each forest polygon (NB: Across the United Kingdom, deprivation cannot be compared directly between devolved nations because distinct methodologies used to quantify it). In England and Wales, income deprivation is reported as the proportion of people experiencing income-related deprivation (Ministry of Housing, Communities and Local Government, 2020). In Scotland, it is the proportion of adults receiving income support, income-based Employment and Support Allowance, or Jobseeker's Allowance (Scottish Government, 2020). In Northern Ireland, it is measured as the proportion of the population living in households where income is below 60 per cent of the national median (NISRA, 2017).

The broad scale trend for the whole of the UK is also reflected at a smaller regional scale. The Northern Forest is an area running from the east to west coast across the North of England, Figure 4. An analysis just within the Northern Forest area, Figure 5, also shows the association between income-related deprivation of a neighbourhood and lower trait richness of the woodlands within that neighbourhood. (Fisher, 2025b, manuscript in preparation).

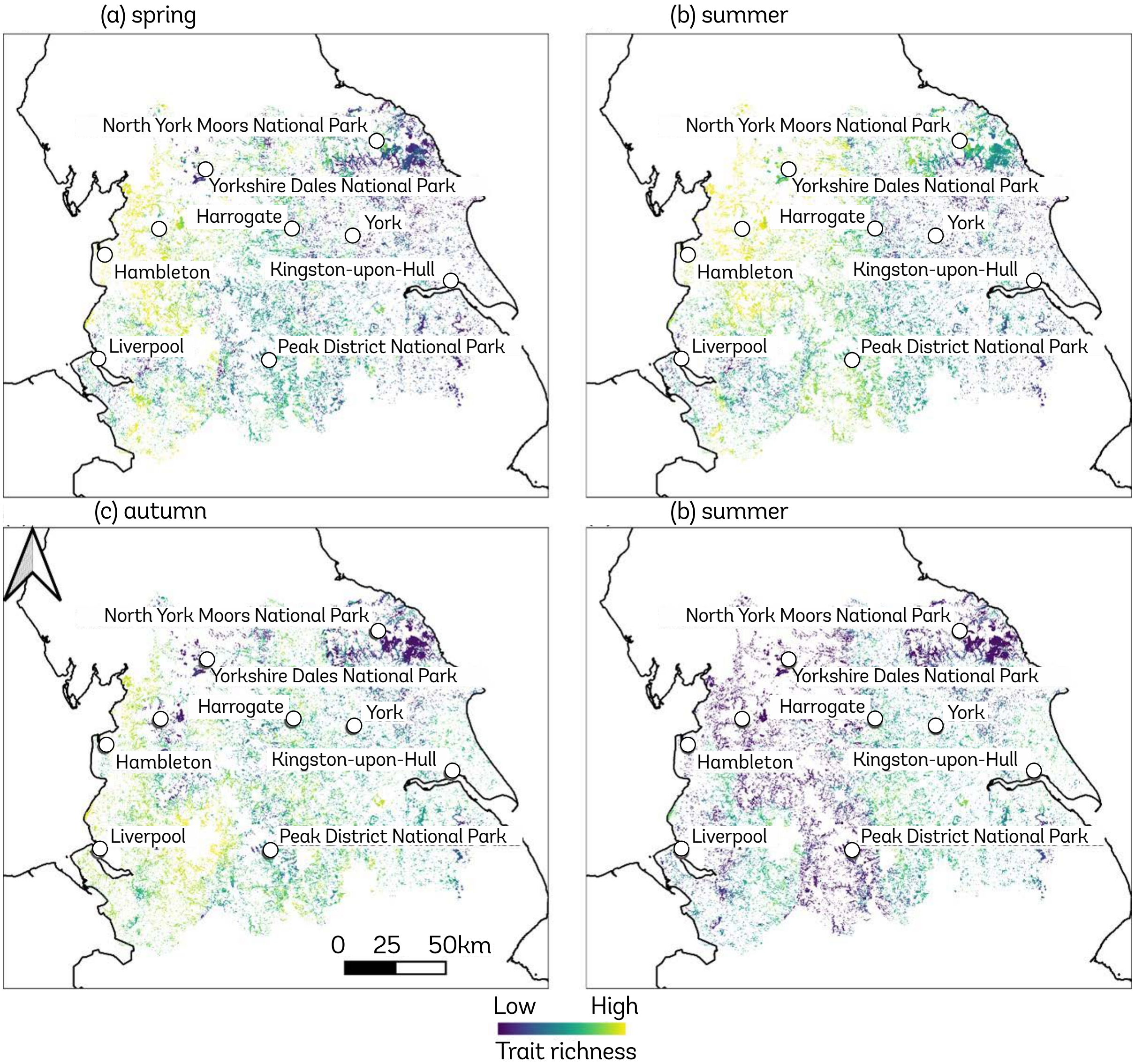

Figure 4. Spatiotemporal patterns of cumulative species' effect trait richness, for the effect traits that underpin positive human wellbeing, across the Northern Forest. Maps depict mean cumulative effect trait richness per forest polygon in the National Forest Inventory (57,093 forests). Cumulative effect trait richness is the total number of unique effect trait-wellbeing incidences across all species, in (a) autumn, (b) winter, (c) spring and summer (Fisher, 2025b, manuscript in preparation).

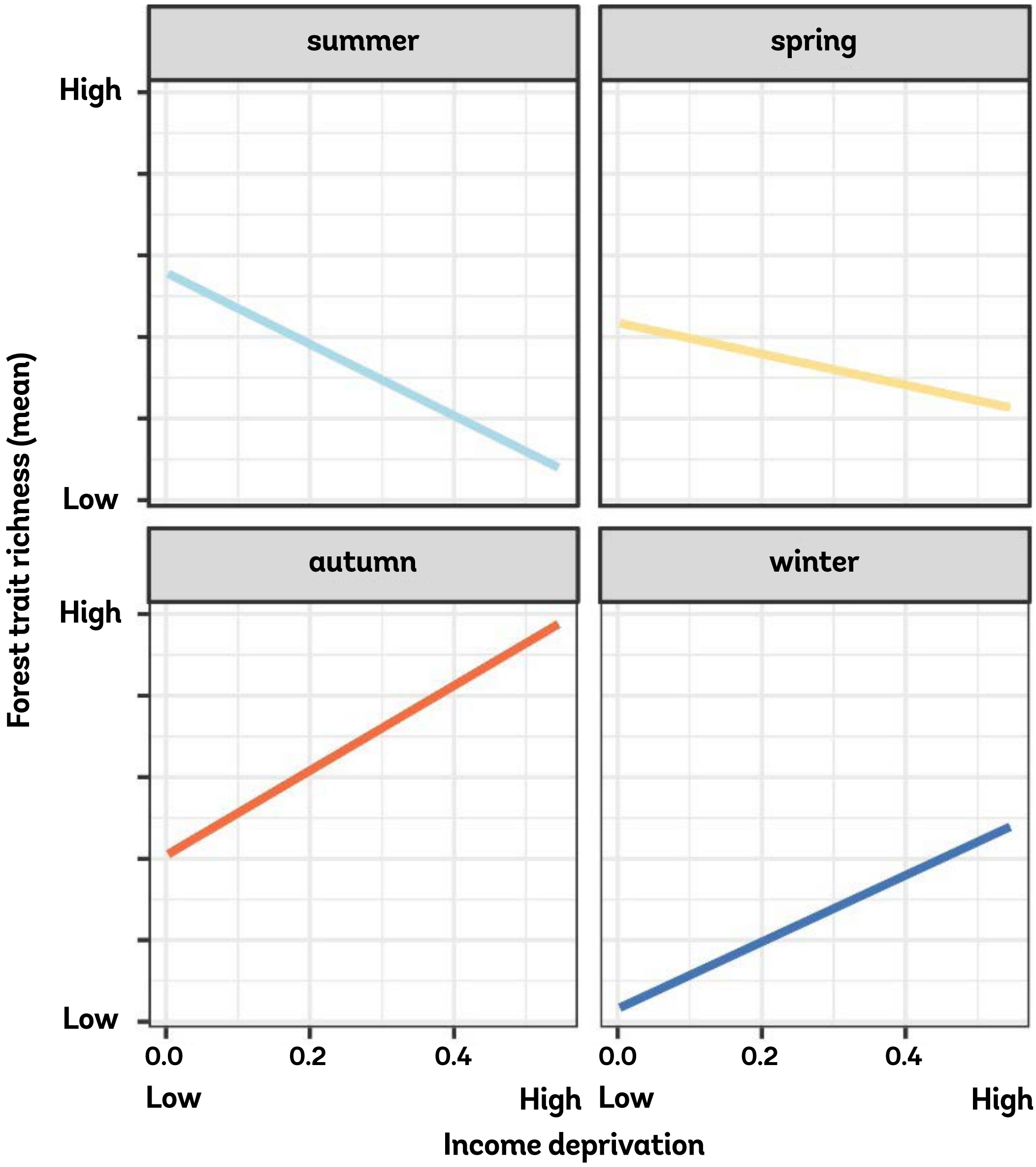

Figure 5. Line graph depicting the linear modelled relationship between species' effect trait richness of forests for positive human wellbeing and income-related deprivation across the Northern Forest for each of the four seasons. Shading around the lines indicates 95% confidence intervals. Deprivation values of the LSOA are extracted at the midpoint of the forest polygon Income deprivation is measured as the proportion of people experiencing income-related deprivation (Ministry of Housing, Communities and Local Government, 2020). (Fisher et al., 2024, preprint; Fisher, 2025a, manuscript in preparation).

These findings expose a new aspect of environmental health inequality: the quality of people's local woods in terms of their ability to deliver wellbeing experiences. In more socioeconomically deprived neighbourhoods, overall environmental health tends to be poor (Evans & Kantrowitz, 2002), and people may be at greater risk for low wellbeing (Castelletti et al., 2024) suggesting these areas should be prioritised for strategically targeted nature recovery.

The relationship found between deprivation and woodland wellbeing-trait richness (Fisher et al., 2024, *preprint;* Fisher, 2025a, *manuscript in preparation*) may somewhat reflect the 'north-south divide' phenomenon in England (Giovannini & Rose, 2020) – the South East of England tends to have less deprivation in general than other parts of the UK. As higher effect trait richness is concentrated in the South East (Fisher et al., 2024, *preprint;*

Fisher, 2025a, *manuscript in preparation*), it is likely this could underpin the general association between the effect trait richness of an area and its level of deprivation. However, even at a smaller scale when the effect of the South East is excluded and the analysis focuses within the Northern Forest landscape only, the same significant relationship is still evident (Fisher, 2025b, *manuscript in preparation*). Within the Northern Forest region with its overall lower trait richness, areas with relatively higher effect trait richness are still related to less deprivation, and lower trait richness associated with more deprivation. That this trend holds, even within a smaller region, raises many questions as to the factors which may underlie or explain this association. Further research into the social, economic, historic and ecological factors behind the trend found here would be needed to better understand how the pattern developed.

Initiatives such as the NHS's Green Social Prescribing (Husk et al., 2020) seek to improve wellbeing by prescribing time in nature. Recent work (Austen et al., 2021; Fisher et al., 2023) showing that people relate to the variety of species and their colours, smells, textures, sounds and behaviours suggests a woodland without wildlife could be likened to a hospital without doctors; the very essence of what makes nature a powerful healer is the range of species it supports. That the richness of potential woodland wellbeing experience is distributed unequally raises questions around whether prescribing activities such as regular woodland walks in one part of the country will be less efficacious as the same prescription for a person living in another region– an uncomfortable possibility which warrants further study.

Conclusion

While woodlands are one of the most visited types of natural environment, the majority of people in the UK don't visit a woodland more than once per month, and many have not visited a woodland in the last few years. While woodland forms only part of the total green and blue infrastructure available and used by the public, this suggests there is potential to increase the number of people benefiting from the wellbeing effects of regularly spending time in woods, as part of broader efforts to increase public benefits from nature in general. Research has provided insight on the barriers which discourage some people from visiting woodlands. Efforts are needed to actively remove or reduce those barriers. There are examples of where such work is underway (Pearson et al., 2023), and evidence to show it can be very effective at engaging a more diverse range of people with woodland-wellbeing experiences (Gittins et al., 2023a; Gittins et al., 2023b)

Investment of public money is improving our understanding of the current state of population-level proximity to publicly accessible woodland in England only. When available, results produced by Forest Research will provide a new baseline allowing woodland access to be tracked more meaningfully into the future but will not allow for time-series comparison with previous analyses based on very different data. Targeted measures are needed to increase provision of woodland access in areas which are currently under-served with greenspace, including woodlands. Research has provided insight on the barriers which discourage some land managers from permitting public access to their woodlands. This improved understanding should now underpin action to support woodland managers by reducing these barriers and therefore making more woodland available for public access.

The differing quality of woodlands may impact their value for wellbeing. Woodland 'quality' can include the quality of human infrastructure available, type and structure of the woodland, and the richness of sensory experience provided by the biodiversity present. Novel research suggests the richness of potential sensory experience from woodland biodiversity may be lowest in the most socioeconomically deprived areas (Fisher et al., 2024, *preprint*; Fisher, 2025a, *manuscript in preparation;* Fisher, 2025b, *manuscript in preparation*). This highlights the importance of enhancing the ecological quality of accessible woodlands, and wider nature recovery at a regional scale, to maximise the wellbeing benefits people can derive from woodland.

Reflections on the importance of woodland access for wellbeing in a changing world

Humanity is facing a suite of unprecedented, interconnected and ever-accelerating global crises. This includes the climate crisis, the biodiversity crisis, and the crisis of deteriorating mental health. All these problems are symptomatic of a broken relationship between human society and the rest of nature (Richardson, 2023). As we navigate the changing world of the Anthropocene, it is essential we build a new cultural connection with nature - as individuals and as a society. This is because a loving relationship with nature drives both pro-environmental behaviour change, and a deep sense of personal wellbeing (Martin et al., 2020).

The UK has one of the least nature-connected populations in Europe (Richardson et al., 2022). To build this new connection will require encouraging and facilitating people's regular engagement with places such as woodlands, rich in the sensory experiences which come with thriving biodiversity.

What needs to be done?

Address barriers which prevent some people visiting woodlands and encourage use of woodlands for wellbeing by diverse publics

Work to encourage and facilitate the use of woodlands by all communities, by overcoming the very real non-physical barriers people experience which prevent their use of woodlands (Pearson et al., 2023). Failure to address these barriers would represent a significant missed opportunity to reduce health and wellbeing inequalities.

This can include improving the quality of woodlands in terms of:

- Infrastructure such as signage, paths, lighting, toilets, disability access.
- Improving inclusivity and welcoming a more diverse range of people through running organised events designed with and for under-represented groups who previously may not have visited woodlands.
- Providing information on where to find woodlands to visit.
- Improving public transport links to woodland sites.
- Improving public awareness and experience of the wellbeing benefits woodlands can offer, through campaigns, education and green social prescribing, celebrity/influencer endorsement etc.

Improve people's proximity to local woodlands in an equitable way

Target interventions to improve provision of woodland access close to the communities who most need more access through:

- Opening existing woodlands for free public enjoyment where possible close to areas of deprivation and where there is little other greenspace provision.
- Targeted creation of new woodlands, especially in any new housing developments and close to more heavily populated, deprived areas where few can currently access any woodland on foot.

Address barriers for woodland managers which discourage provision of public access to woodlands

Little evidence has evaluated the impact of existing interventions to support land managers to increase public access provision. However, the evidence review by McConnachie and Gardner (2024) suggests successful interventions will need to:

- Appeal to land managers' values, motives and objectives.
- Provide sufficient economic support.
- Support woodland managers to provide visitor information which educates the public.
- Support land managers to manage ongoing public access.

Improve quality of woodlands in terms of the richness of sensory and cultural wellbeing experiences they can provide, through enhancing biodiversity

Landscape-scale nature recovery should be considered an essential enabling condition for any plan to improve wellbeing for people through access to woodlands, recognising the essential role of biodiversity in the ability of woodlands to provide wellbeing effects.

Landscape-scale recovery of woody habitats should be targeted to regions which are most depleted of the woodland biodiversity which brings wellbeing, and where the socioeconomic status of the area suggests local people could experience particularly strong benefits in terms of improvements to wellbeing. As discussed in other areas of this report this requires improvement to the extent, condition and connectivity of wooded habitats.

What further evidence is needed?

Continue regular data collection on public visiting rates to woodlands

Continue surveys with consistent methodologies and sample sizes to understand changes in people's frequency of woodland visits into the future. This is essential to enable progress-tracking towards increasing the actual use of woodlands by the public. Data on visiting rates broken down by demographic categories would help understand whether progress is being made in making woodlands more inclusive places.

Produce an up-to-date estimate for the proportion of the population meeting the Woodland Access Standards in Northern Ireland, Wales and Scotland.

In Wales, the dataset on the locations of publicly accessible woodlands has not received a comprehensive update. Investment into improving data on the location of accessible woodland in Wales is needed.

In Northern Ireland, the GreenspaceNI Map could be used to produce a woodland-focused estimate for public access. These spatial datasets will require long-term maintenance to ensure its quality is upheld into the future.

For Scotland, all woodland is publicly accessible, so a special 'accessible woodland' dataset is not required. The National Forest Inventory dataset can be used to produce an estimate of how many people meet the Woodland Access Standard in Scotland.

Long-term active maintenance of data on publicly accessible woodlands

Well-maintained datasets on public access to woodlands across all four UK countries are required to enable targeted interventions to improve woodland access. In England, the policy commitment to deliver a minimum level of access to greenspace through the Environmental Improvement Plan led to resources being focussed toward better understanding access to woodlands (along with other green and blue spaces) within the country. The Forestry Commission produced the 'Woods for All' dataset on the location of publicly accessible woodland. DEFRA has produced an analysis of overall greenspace access and Forest Research is producing an equivalent baseline for woodland access (as yet unpublished at time of writing). Looking forward, future updates to the dataset and re-analyses at strategic time points will need to be considered.

Learnings from the previous two decades since the initial creation of the Woods for People dataset show the importance of continued investment of sufficient effort and resource to maintain the quality of woodland access data throughout the coming decades. Investment in data on the location of publicly accessible woodlands should reflect the great value of this data for society.

Research to better understand the wellbeing effect trait richness map

The wellbeing effect trait richness map (Fisher et al., 2024, *preprint;* Fisher, 2025a, *manuscript in preparation*) presented in this report explores wellbeing from nature in a completely new way. It suggests hypothetically there may be a difference in the strength of wellbeing benefit a person would gain from visiting woodlands with different levels of predicted effect trait richness. Further research to quantify in-situ wellbeing benefits from users of woods with differing levels of predicted effect trait richness could help verify whether predicted trait richness is related to in-situ wellbeing benefits.

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Woodland biodiversity

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Abstract

Drawing on the recent publication of the Bunce survey (Smart et al., 2024), Joint Nature Conservation Committee (JNCC) biodiversity indicator data and individual 'State of' reports, we explore population trends for woodland-associated taxa and the potential drivers of change. The results of the Bunce survey indicate a general trend towards fewer, older and larger trees within the woods sampled and a reduction in open habitats and micro-habitats. The survey also highlights how certain floral species are benefiting from these changes while others are not. Trends in JNCC species indicators and 'State of' reports are generally consistent with this trend of woodland development. We discuss the implications of the findings outlined above for British woodlands and the impact of climate change, and how we can increase woodland resilience by building in complexity via enhanced conservation management.

Introduction

Biodiversity can be defined as 'the variety of life on Earth, in all its forms, from genes and bacteria to entire ecosystems' (United Nations, 2022). Woodlands therefore both comprise and are components of biodiversity, and individual tree, shrub and wildflower species are as much a part of biodiversity as the birds, bees and butterflies more commonly referred to as wildlife. Each species has its own requirements and is constantly responding to the changing world around it. In this section we will explore how different woodland species are faring. Which species are thriving? Which species are declining? How might different species respond to a changing world and how will this affect woodlands over time?

Earth is currently experiencing a period of intense change, with human activity driving a sixth mass extinction through a combination of habitat destruction, movement of invasive species, overexploitation, illegal wildlife trade, pollution and climate change (IUCN, 2022). Emissions released by the burning of fossil fuels are drastically altering the climate, leading to record global temperatures and increased prevalence of extreme weather events such as flooding and drought (Met Office, 2018). While climate change exerts obvious pressures on individual species across the globe, affected biological communities may themselves accelerate climate change if they become degraded to the point that the amount of carbon they are able to absorb is affected (Natural History Museum, 2022), or as has been observed in certain ecosystems, they become net emitters of carbon (Gatti et al., 2021).

Although the biodiversity and climate crises are global in nature, some areas are more affected than others. Britain is one of the most nature-depleted countries on earth and is forecast to experience a range of climate change effects including warmer summers, wetter winters and increased frequency and severity of heatwaves, droughts and flooding events (Met Office, 2020). It is therefore essential that native species and habitats are able to adapt to a changing climate, both to conserve them in their own right and to create and maintain landscapes which are resilient to climate change.

Woodlands support an abundance of biodiversity, but woodland cover remains low compared to other countries in Northern Europe and many woodland-associated species in Britain continue to experience long-term declines (Reid et al., 2021). Here, we provide the latest assessment of how woodland biodiversity in Britain is faring at the species level in the context of the global biodiversity and climate crises. This is essential for prioritising conservation efforts, as is identifying the extent to which different environmental drivers are influencing species population trends. (Implications for genetic-level diversity are considered in the context of tree regeneration in the woodland condition and tree provenance sections of the report).

Long-term monitoring data is required to enable these analyses, and one such long-term monitoring initiative is the Bunce Woodland Survey of Great Britain, hereafter referred to as the Bunce survey. The Bunce survey (Smart et al., 2024) comprises three separate surveys undertaken in 1971, 2001 and 2021, and provides unparalleled insight into how Britain's broadleaved woodlands have changed over the last 50 years and some of the major drivers of these changes.

We also report on the most recent JNCC biodiversity indicators and various species-focused 'State of' reports which provide vital insight into how we can improve the condition of woodlands in Britain for woodland biodiversity. By tracking species population trends and seeking to understand their causes, we are better equipped to reverse the declines in woodland biodiversity and offer holistic, landscape-scale solutions to provide the conditions necessary for biodiversity to thrive.

Methods

We summarised the findings of the Bunce report, which analyses 50 years of data from the 1971, 2001 and 2021 Bunce surveys, JNCC biodiversity indicator data and individual species-focused 'State of' reports (namely, 'The State of Britain's Dormice 2023' and 'The State of the UK's Butterflies 2022', and country-level 'State of' reports published by the State of Nature Partnership in 2023). It is important to note that the latest round of the Bunce survey includes data from 97 broadleaved woodland sites (each of which includes 16 survey plots) which are broadly representative of the broadleaved woodland types found in Britain. As such, it has been possible to make inferences about broadleaved woodlands across Britain by using the data in the Bunce report. In each woodland plot, observations were made of the composition and frequency of tree and shrub canopy species, the presence and cover of understorey plant species, and plot attributes such as presence of micro-habitats and signs of grazing. Soil samples were also taken. Using the data collected during these observations, response variables were developed which included ground flora species richness and species diversity, tree species richness and measurements, and soil properties. Statistical analyses were then undertaken to allow for comparison of response variables both across sites and surveys. Historical climate and atmospheric deposition data were used in the analyses, and categorisation of deer risk for different sites was based on expert opinion and observations. For a more detailed explanation of the full survey methodology please see the Bunce report (Smart et al., 2024).

Results

Findings from the Bunce report

How has woodland structure changed over time?

While the frequency of the most common tree species making up the woodland canopy remained stable between 1971 and 2021 in the Bunce survey, there have been significant changes in woodland structure over time.

Woodlands in 2021 are shadier than in both 1971 and 2001, owing to a combination of natural stand succession, whereby the young and open woodlands surveyed in 1971 have over time developed into woodlands containing older and fewer trees, and a decline of management interventions which create canopy gaps. In the 1971 and 2001 Bunce surveys, signs of recent canopy gap creation outside forestry estate woods were observed in around 20% of survey plots. In 2021, this was closer to 10%. While gaps as a result of tree disease and storms were increasingly evident in the most recent survey, their occurrence has been outpaced by the general trend of canopy closure. Additionally, microhabitat diversity, the diversity in small-scale habitats which differ from their surroundings in some way, decreased over the 50-year survey period.

As a result of this trend of stand development and decline in deliberate management (in the form of canopy gap creation), the surveyed woodlands now contain 60% fewer trees than in 1971 while mean basal area has increased.

These changes in structural composition have far-reaching consequences for woodland biodiversity. For example, the amount of light which reaches the woodland floor is reduced under denser canopies, which will affect ground flora in various ways depending on whether they are light loving or shade loving. The response of plant communities to widespread changes in light levels is also likely to have knock-on impacts on other taxa that depend on certain plant species for foraging and nesting, amongst other things.

How are woodland ground flora responding to changes in light levels, soil pH and herbivory?

Ground flora species richness - the total number of species in a given area - declined significantly in the Bunce plots between 1971 and 2001 and then increased significantly between 2001 and 2022. However, in 2021, species richness was still significantly lower than in 1971, resulting in a net 22% reduction over the 50-year period (see Fig. 1). It is important to note that while species richness does not necessarily correlate with conservation value, it can be a good indicator of habitat quality and confer greater resilience to habitats and ecosystems (Bullock et al., 2022).

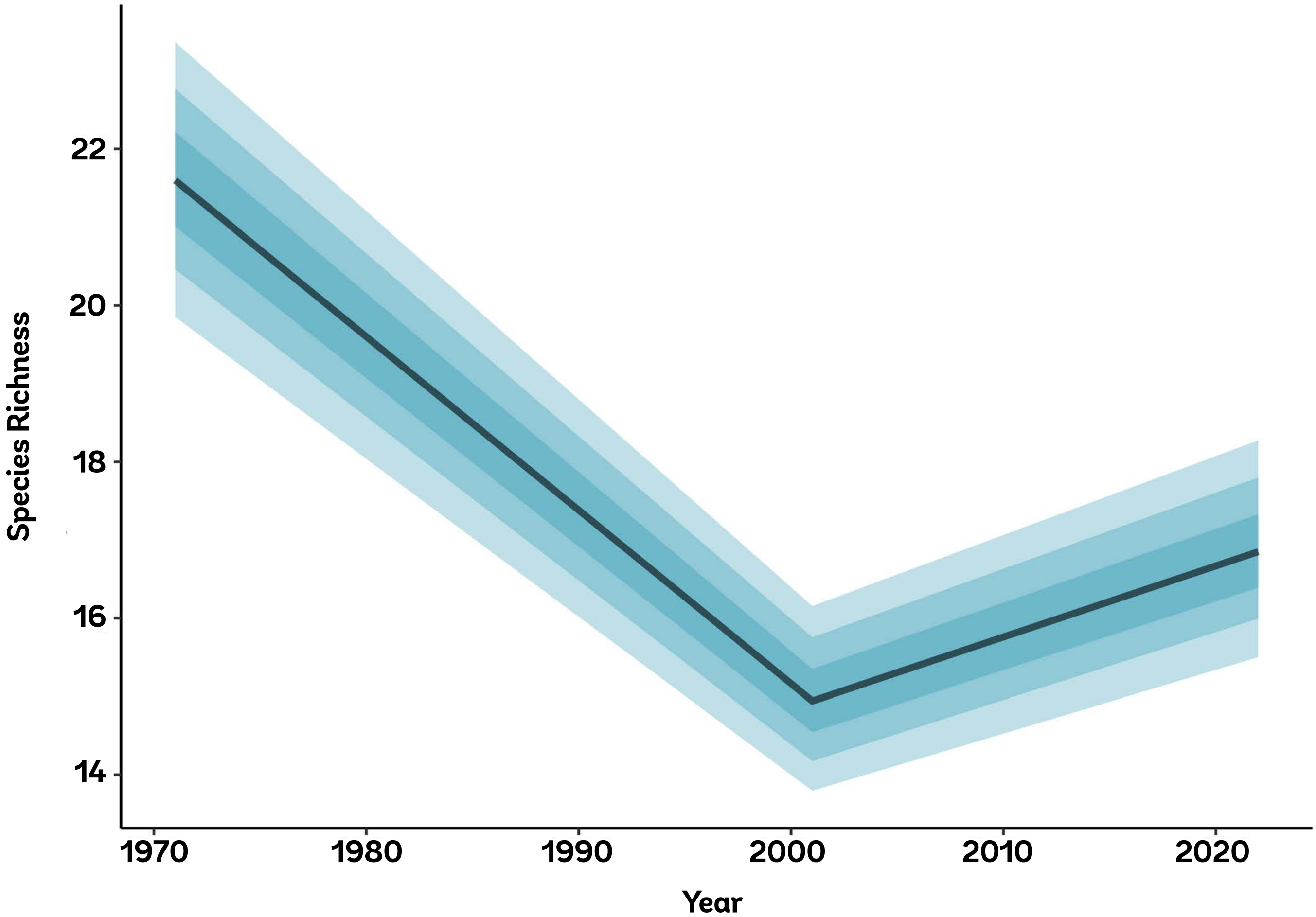

Figure 1: Change in plant species richness having accounted for a mean effect of day-difference in survey dates between each survey. Taken from Smart et al. (2024).

Figure 1 displays the change in mean species richness across the survey period. As changes in woodland structure and composition affect different species in different ways, changes in mean species richness can mask trends for individual species. For example, shade tolerant species such as buckler ferns (*Dryopteris* spp.) and hart's-tongue fern (*Asplenium scolopendrium*) , remote sedge (*Carex remota*), and holly (*Ilex aquifolium*) seedlings, increased in abundance (either mean percentage cover or frequency of plots they were present in) across the survey period, while light-demanding species such as heath bedstraw (*Galium saxatile*) and rosebay willowherb (*Chamaenerion angustifolium*) have shown decreases in abundance (frequency of occurrence for heath bedstraw and both frequency of occurrence and mean percentage cover for rosebay willowherb).

As species richness declined from 1971 to 2001, and increased from 2001 to 2021, the proportion of ancient woodland indicator species remained relatively constant (42%, 42% and 43% of the mean richness in plots in 1971, 2001 and 2021 respectively). However, ancient woodland indicators such as wild garlic (*Allium ursinum*), dog's mercury (*Mercurialis perennis*) and bluebell (*Hyacinthoides non-scripta*) either remained constant or increased in cover between 1971 and 2021. All three species are perennials capable of exploiting stable, shaded conditions.

Likewise, the mean proportion of non-woodland specialist plants remained stable at an average 26%, 28% and 28% mean richness in 1971, 2001 and 2021.

But while the proportion of non-woodland specialist plants has remained stable, these species have become more frequent in plots associated with canopy gaps and less frequent in plots under the canopy. Taken together, these results suggest that the changes in overall species richness across the survey period have not been selective for either woodland specialists or generalist plants, but rather for those species tolerant of shade and clonal or large woodland specialist flora capable of increasing in cover in the stable conditions provided by mature woodlands with few canopy gaps. Smaller woodland plants such as wild strawberry (*Fragaria vesca*) and wood sorrel (*Oxalis acetosella*) displayed decreases in abundance over the survey period.

In addition to shade tolerant species, another group of species which appear to be increasing in abundance in the Bunce plots are those of more fertile conditions. Species such as cleavers (*Galium aparine*), common nettle (*Urtica dioica*), dandelion (*Taraxacum* agg.) and perennial rye grass (*Lolium perenne*) all increased in abundance throughout the survey period. It is possible that the apparent selection for nutrient-loving species is a result of reduced woodland management and the resulting reduced offtake of nutrients, in addition to external inputs from nitrogen deposition and nutrient surplus from surrounding land. It is also possible that recovering soil pH levels throughout the survey period as a result of falling sulphur deposition levels may be driving an enhanced eutrophication response in areas of high nitrogen deposition by increasing macronutrient availability.

Previous studies have shown that light limitation associated with closing canopies can suppress the response of shade-intolerant nitrophilous species (Smart et al., 2014), although no relationship between woodland gaps and mean Ellenberg N (a measure of a plant's affinity for nitrogen) was observed in the Bunce survey. Changes in light levels in these gaps may be influential in determining how their species composition and mean Ellenberg N alters over time.

Pressure from herbivory also influences woodland understories. While signs of livestock and red deer reduced slightly from 1971, signs of other deer species increased over time in keeping with the known increases in their numbers across Britain. Deer preferentially browse or graze palatable species which creates different levels of pressure on different plant species. This has important implications for woodland plant communities, with previous studies (Morecroft et al., 2001) finding that high deer pressure can favour grass species and decrease forb abundance. The results of the Bunce survey indicate a slight reduction in bramble (*Rubus fruticosus* agg.) cover and a doubling of grass cover in high-risk deer grazing sites from 1971 to 2021, while a doubling in bramble cover and a halving of grass cover was observed in low-risk sites (Seaton et al., 2024). The results also indicate that grass species of grazed and better lit conditions, such as common bent (*Agrostis cappillaris*) and sheep's fescue (*Festuca ovina*), decreased in abundance across the survey period.

How is climate change affecting woodland flora?

Over the survey period, mean tree DBH (diameter at breast height) in the Bunce plots was positively correlated with increase in summer maximum temperature. However, mean tree DBH increased less at sites that experienced higher maximum summer temperatures, suggesting that trees on these sites may already experience limiting factors such as drought.

At the individual species level, different species will react differently to warming conditions, which was observed in the data. One species which showed a large increase in prevalence across the survey sites over the survey period is holly, a shade tolerator which has now overtaken hawthorn and beech in terms of plot-frequency (the proportion of plots with holly present). It is now found in 16% more plots than in 1971. In contrast to other tree and shrub species which generally lost stems in younger age classes throughout the survey period, holly had more young stems in 2001 and 2021 than in 1971.

Rising winter temperature was identified as a main driver of the increase in occurrence of holly. Plot-frequency of holly was positively correlated with change in winter minimum temperature, but the increase was greater at sites that started out warmer. This was true for all age classes for the species.

In summary, holly is increasing in plot-frequency, but it is increasing the most in sites with warmer winters. This demonstrates one way in which the composition of woodlands is changing due to climate change, and how this effect can differ in different areas of UK. Holly is unlikely to be a unique case. It is possible that many other species will be similarly affected by warmer winters, or will begin to be, as temperatures and other climatic factors continue to change.

One group of species which is responding positively to a warming climate is those with a southerly distribution (i.e those species in a given area which are at the northern limit of their current range), such as common ivy (*Hedera helix),* pendulous sedge (*Carex pendula*) and soft shield fern (*Polystichum setiferum*). The proportion of southerly distributed species recorded in survey plots increased between all survey years, with this proportion being, on average, higher if sites had become warmer in the summer. However, although it appears those species with a southerly distribution became more widespread throughout the survey period, it seems that the beneficiaries of these changing climatic conditions comprise those plants which are capable of existing in shade. Species including common bird's-foot-trefoil (*Lotus corniculatus*), cock's-foot (*Dactylis glomerata*), foxglove (*Digitalis purpurea*) and Yorkshire-fog (*Holcus lanatus*), which are all shade intolerant species, declined over the survey period. Although it appears that there has been clear selection for species tolerant of shade, southerly distributed species still comprise a greater proportion of plot species richness than they did in 1971 and 2001 suggesting that woodland composition is changing as a result of a warming climate.

Additional implications for specific species

Canopy gaps in the surveyed woodlands were less frequent than ever in 2021, but where present, can support those species not capable of surviving under shady closed canopy conditions and increase overall species richness. Recent storm damage was recorded in 5% of plots in the 2021 survey and acts as a mechanism for creating openings in the canopy and gaps for light to penetrate. The death or dieback of trees from pests or diseases is another mechanism of gap creation in woodlands. One such disease that was first recorded in the UK in 2012 and has spread rapidly with high mortality rates is ash dieback (*Hymenoscyphus fraxineus*). It can lead to the death of ash canopies and entire trees and stands of trees. The spread of this disease and its effects on woods comprising ash are evident in the Bunce data which predate and precede the arrival of the disease.

In the Bunce woodlands, ash was present in 44% of the plots surveyed in 2021, with signs of ash dieback observed in 49% of these plots. In total, ash dieback was observed in 21% of all plots. Plant species richness was higher in plots with ash dieback than plots without, while plots without any ash trees showed lower species richness still. Relationships between deer browsing levels and associated grass, forb and bramble cover were also detected in the data, whereby plant species richness has only increased in plots with ash dieback in areas of high deer grazing. Forb cover on the other hand, increased in plots with ash dieback under all levels of deer grazing, suggesting that changes in species richness associated with ash dieback and deer grazing are due to deer preventing the dominance of certain forbs species, such as bramble.

As ash dieback continues to spread throughout the UK, canopy gaps will become more common for a period, and it will be important to monitor how populations of light-demanding and nutrient-loving species adapt to the changes in light regimes. Over time, these gaps are likely to be filled by recruits of other species (Needham et al., 2016), and in the absence of conservation management or disturbance will eventually revert to a closed canopy, which may or may not be desirable depending on conservation objectives. The tree species which colonise and dominate these gaps are also likely to have far-reaching implications for overall species richness. Sycamore saplings for example, are considered most likely to replace ash in most of Britain, and while sycamore may be a good candidate to replace ash based on the amount of ash-associated biodiversity it supports (Mitchell et al., 2014), it casts a deep shade and is unlikely to support the same floral species richness as ash, which provides light-dappled shade suitable for many woodland ground flora species. In addition to the potential implications of a changing light regime associated with a decline in ash, ash itself was found to support 1,058 species, 44 of which are 'obligate' ash-associated species, only found on living or dead ash trees (Mitchell et al., 2014). Taken together, these results highlight the effect the loss of a single woodland keystone species can have on woodland communities.

Trends in species indicators

In addition to the Bunce survey, which detects broad trends in woodland vegetation composition and structure in a sample of British woodlands, species population data from long-term national monitoring schemes further contribute to our understanding of how woodlands and the species that inhabit them are changing over time.

Taxon-specific data is often converted into an index, a statistical measurement of change over time, and reported on as part of the UK Biodiversity Indicators. The data is collected for taxon-specific recording schemes, coordinated by a range of different organisations.

Here we will examine biodiversity indices for woodland plants, birds and butterflies, which were reported on in the previous State of the UK's Woods and Trees report, in addition to mammals, as these species are widely recorded and provide sufficient data to inform statistically robust indices.

Plants

The Botanical Society of Britain and Ireland's Plant Atlas 2020 (BSBI, 2021) provides an insight into how our native plant species' distributions are changing. It uses hectad (10km by 10km squares) data to compare species presence over the long term (1930 to 2019) and the short term (1987 to 2019). Due to differences in recording resolution, it is difficult to compare the results of the Plant Atlas directly with the Bunce survey, although the results appear to be consistent. While the distributions of woodland plants have remained relatively stable, some specialists of more open areas such as early purple orchid (*Orchis mascula*) and oxlip (*Primula elatior*) have declined while shade-tolerant species such as broad buckler fern (*Dryopteris dilatate*) and male fern (*Dryopteris filix-mas*) have increased. Species associated with conifer woodland have increased due to the expansion in plantation forestry. When considering the results for all habitats, the findings also show declines in those species adapted to infertile conditions and low competition, and northward range expansion of certain southern species (Walker et al., 2020). The National Plant Monitoring Scheme (NPMS, 2015) is another large-scale plant monitoring programme that can help us understand how woodland plant populations are changing. The JNCC's Plants of the Wider Countryside index (JNCC, 2023a) is an experimental statistic (i.e. not currently an official statistic within the UK Biodiversity Indicator set) that utilises NPMS data to summarise the modelled percentage cover of a set of plant species indicative of good habitat condition.

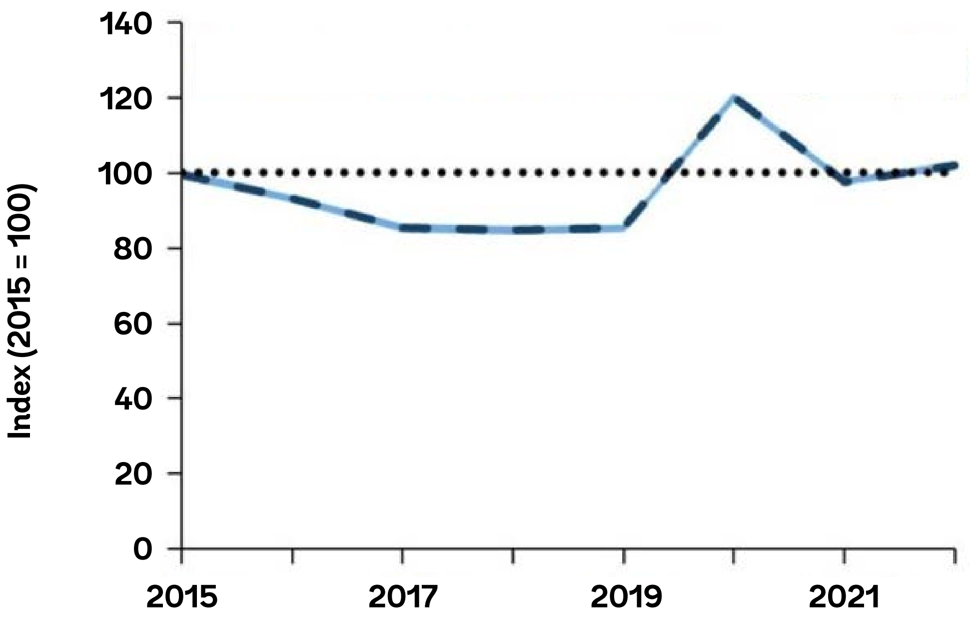

Figure 2. Abundance of plant species in four UK broad habitat types, 2015 to 2022)

After declining by 18% between 2015 and 2019, the Plants of the Wider Countryside index has since risen and is now 2% above its 2015 level (as shown in Figure 2), displaying a similar trend to other broad habitat types. However, as this statistical model is still classed as experimental and features data from relatively few years, it is difficult to draw confident conclusions on the long-term trends. Trends for the average cover of individual species which contribute to the indicator can be viewed on the graph on the previous page (NPMS, 2024), most of which are broadly stable.

Birds

Birds have well established monitoring schemes and make good indicators as they occupy a range of habitat niches and respond quickly to changes in their environment. However, using birds as indicator species also comes with limitations. They are often highly mobile and, in many cases, migratory, and may use more than one habitat or breeding or wintering ground, making interpretation of results difficult. They are also susceptible to a range of causes of mortality, including direct persecution, which isn't related to their habitat or environmental drivers.

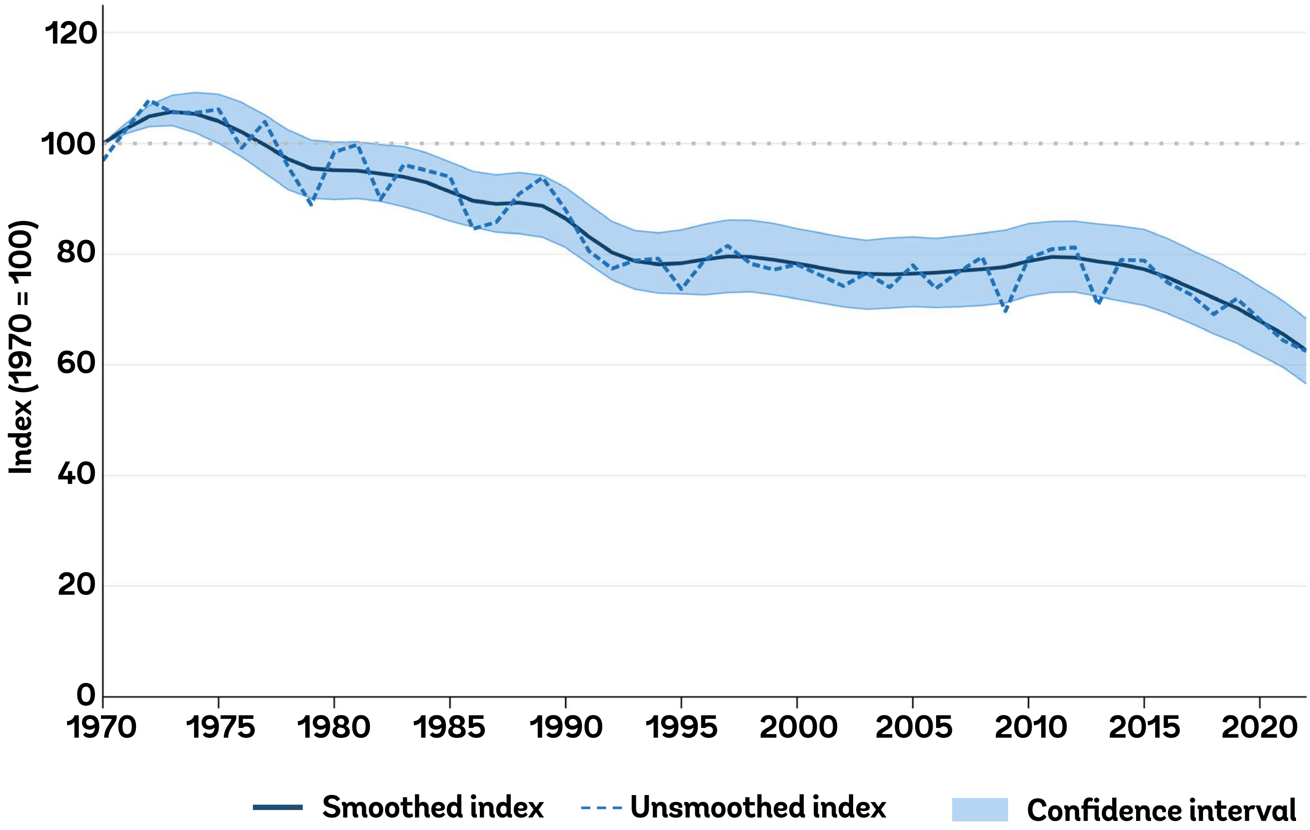

Figure 3. Breeding woodland birds in the UK, 1970 to 2022. Source: JNCC (2023b) with permission from JNCC.

The JNCC woodland bird index (as shown in Figure 3) (JNCC, 2023b), which draws on data from 37 woodland species, was 37% lower in 2022 than in 1970, and in the last five years has decreased by 15%. In the short term (2016 to 2021), species showing strong or weak population declines comprise 32% and 30% of all species included in the index respectively, compared with 24% and 8% over the long term (1970 to 2021). Additionally, species increasing in abundance make up less of the total species pool in the short-term index.

Species indices must be used with caution, however, as using a combined species index in this way can mask trends for individual species or types of species. Splitting the bird index into woodland specialists versus generalists showed that in 2022 the woodland specialists index was 55% lower than in 1970, while the index for woodland generalists was only 5% lower than in 1970. This is not to say that all woodland generalists are coping with a changing environment. Bullfinch (*Pyrrhula pyrrhula*) and song thrush (*Turdus philomelos*), two species which use a wide range of habitats, have declined by more than 45% since 1970, while long-tailed tit (*Aegithalos caudatus*), another woodland generalist, has increased by 90% in the same period. These disparities in fortune highlight the need to treat these indices with caution when attempting to infer species specific trends.

Woodland specialists, which have shown more severe declines since 1970, display similar disparities. Lesser spotted woodpecker (*Dryobates minor*), spotted flycatcher (*Muscicapa striata*), capercaillie (*Tetrao urogallus*) and willow tit (*Poecile montanus*) have declined by over 90%, while other species have shown significant increases. Nuthatch (*Sitta europaea*) has trebled its population since 1970, while blackcap (*Sylvia atricapilla*) and greater spotted woodpecker (*Dendrocopos major*) have more than quadrupled.

Where available, country-specific population trends paint an interesting picture. England's woodland specialist index (DEFRA, 2024) was 51% lower than in 2022 than in 1970, while the woodland generalist index was 10% lower. Woodland specialists decreased by 17% between 2016 and 2021 while woodland generalists decreased by 12%. In Scotland, the woodland bird index increased by 56% between 1994 and 2022 (Nature Scot, 2024). Birds such as blackcap, greater spotted woodpecker and tree pipit (*Anthus trivialis*) have increased in numbers, while woodland generalists like bullfinch and song thrush, which are struggling at a UK level, are increasing in Scotland. A changing climate may be driving certain changes. For example, willow warbler (*Phylloscopus trochilus*) populations have increased in Scotland and its range has shifted northwards, but most of England has now exceeded the optimum breeding season temperature for the species (Martray et al., 2023). Capercaillie, which has previously been linked with a negative effect of rising temperatures (Moss et al., 2001), has declined by 76% since 1994.

Data for Wales and Northern Ireland is scarcer, but general trends are available. In Wales, woodland birds increased by 33% between 1995 and 2020, although the proportion of generalist species may be driving this overall increase (State of Nature Partnership, 2023a). In Northern Ireland, however, woodland birds declined by 18% between 1996 and 2021 (State of Nature Partnership, 2023b).

While the country-level differences in woodland bird populations may reflect slight differences in recording methodologies, it is likely that country specific drivers also play a role. Understanding these differences will be key to preserving woodland bird populations. Further information on country-specific population trends can be found on the British Trust for Ornithology's Bird Trends Report and Bird Trends Explorer (British Trust for Ornithology, 2020).

Mammals

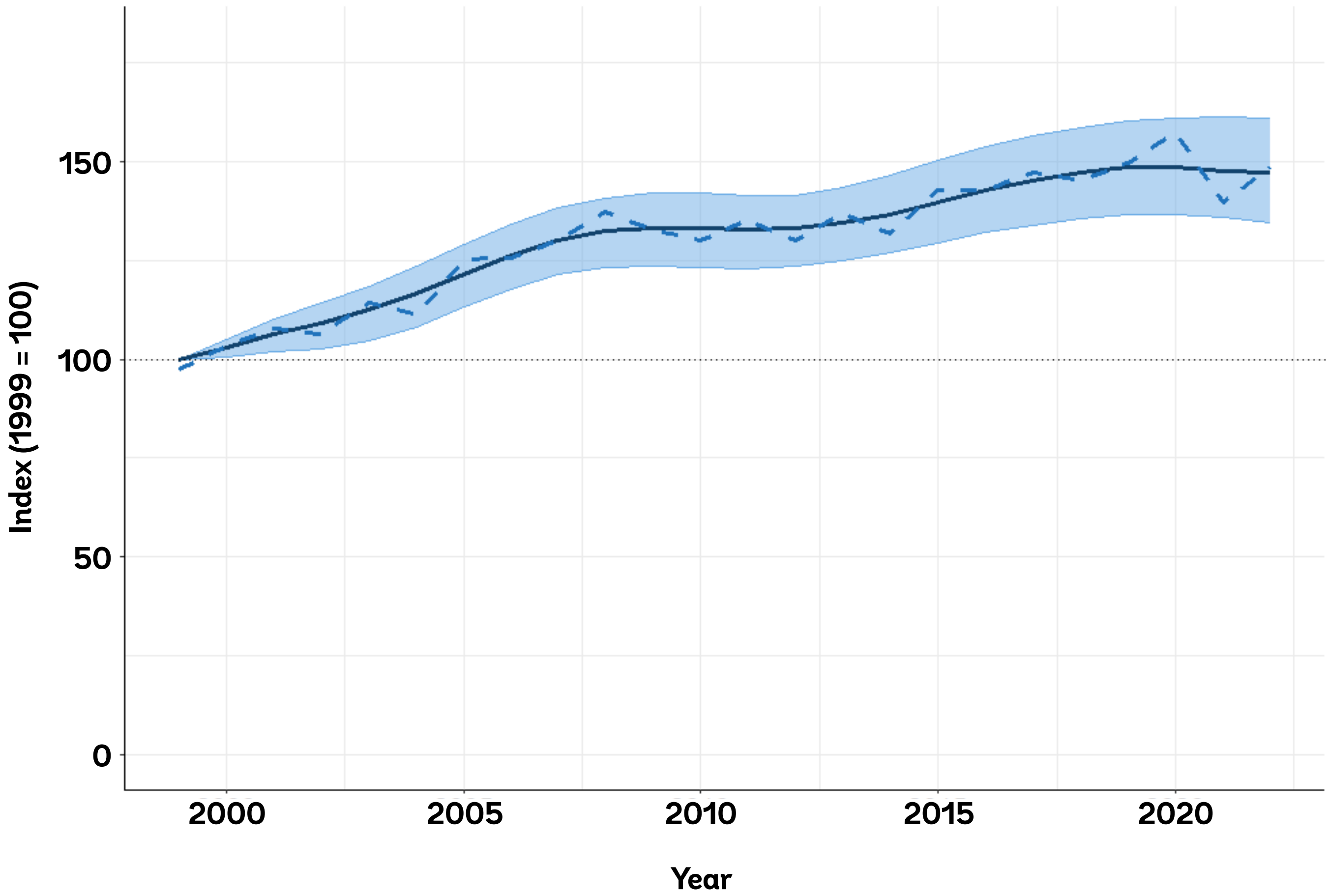

Figure 4. Trends in 11 bat species' populations, in the United Kingdom, 1999 to 2022. Source: JNCC (2023c) with permission form JNCC

The JNCC bat index (as shown in Figure 4) (JNCC, 2023c), which contains data for 11 species, increased by 49% between 1999 and 2020, and by 6% between 2015 and 2020. However, as with the bird indices discussed above, composite indices like these can mask individual species trends. Here, the overall increase in the index over the long term is driven by population changes in three species, including the lesser horseshoe (*Rhinolophus hipposideros*), a woodland specialist species which has increased by 119% since 1999. Another woodland specialist species, Natterer's bat (*Myotis nattereri*), has shown a weaker but nonetheless detectable increase over the same period.

There is evidence that some bat species are beginning to recover from significant historical declines, with the reduction of disturbance since the introduction of legal protection and a milder climate having a positive impact (Burns et al., 2016). Climate changes over winter and spring have also been shown to benefit horseshoe bat species (Schofield, 2008), although the impact of climate on other bat species is less clear.

Country level changes largely reflect what is recorded at UK level. In Wales, an abundance index comprising six species increased by 76% between 1998 and 2021 (State of Nature Partnership, 2023a), while in Northern Ireland a similar index increased by 55% between 2003 and 2021 (State of Nature Partnership, 2023b). For England and Scotland, generic mammal indices include signs of recovering bat populations (State of Nature Partnership, 2023c, 2023d).

Another mammal which is found in our woodlands and is well recorded is the hazel dormouse. Dormice occur in a wide range of habitats including hedgerows, scrub, road and rail verges and even conifer woodlands, but their optimal habitat is early successional woodland featuring a mix of young and more established shrubs and trees.

The National Dormouse Monitoring Programme (NDMP) (People's Trust for Endangered Species, 2019) collects nestbox data from hundreds of woodland sites for annual population assessments. Between 2000 and 2022, results from the programme show that the population of dormice in surveyed woodland fell by 70% (Wembridge et al., 2023), although there are signs that the decline isn't accelerating. If the current trend continues, dormice populations will have fallen by 94% from 2000 levels in just 30 years. Using data collected by the NDMP, Goodwin et al. (Goodwin et al., 2018) have demonstrated that while factors such as climate, location and woodland size can all affect dormouse breeding success and abundance, the influence of habitat and woodland condition together outweigh the effect of climate, with greater dormouse abundance and higher breeding rates being associated with active woodland management. Dormouse were also found to be less abundant on sites with local climates characterised by warmer, more variable winter temperatures, which is likely to have important implications for the species in the face of rising global temperatures due to reduced hibernation survival rates.

Invertebrates

Woodlands support a huge variety of invertebrate species; however reliable population trend data is scarce for many groups. One group which is subject to long-term monitoring programmes is the Lepidoptera, or butterflies and moths. Butterflies in particular are highly visible and make great indicator species as they react quickly to environmental changes. They often display limited dispersal ability and foodplant specialisation in addition to close reliance on weather and climate.

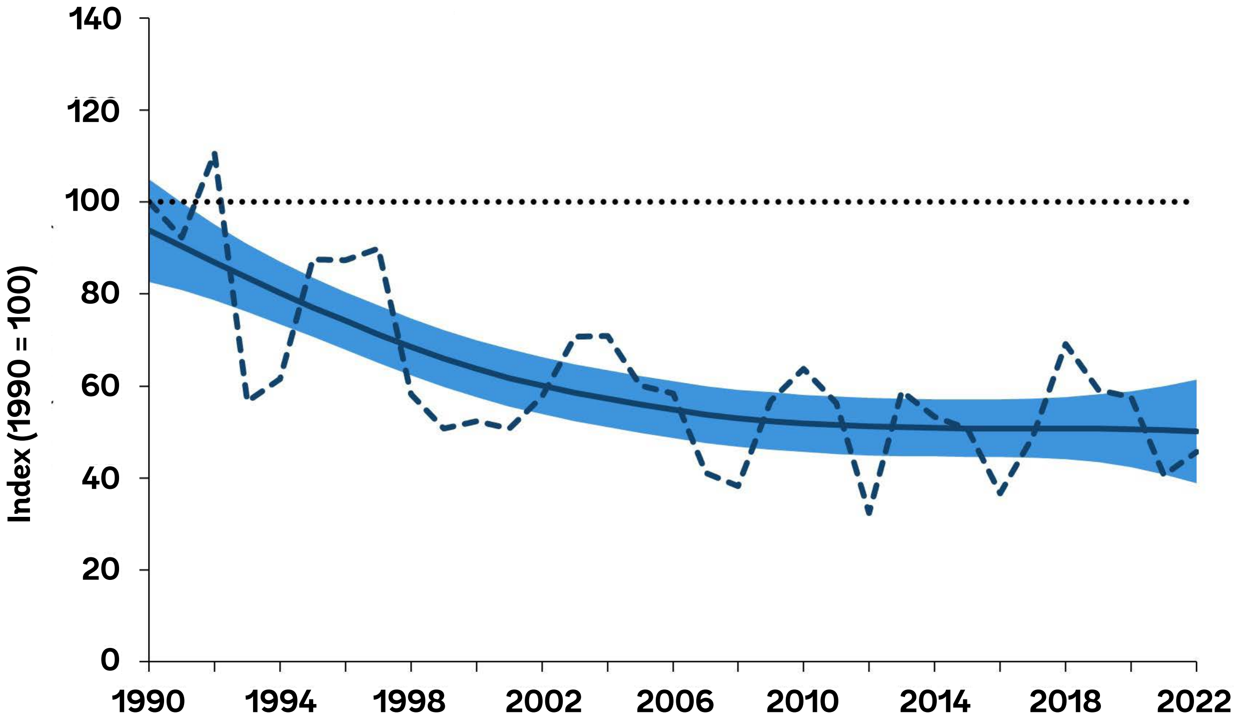

Figure 5. Trends for butterflies of the wider countryside in UK woodland, 1990 to 2022. Source: JNCC (2023d) with permission from JNCC

Large fluctuations in numbers between years are a typical feature of butterfly populations, but by monitoring trends over time, a pattern can be observed. The JNCC 'all-species' butterfly index declined by 15% between 1976 and 2022 (JNCC, 2023d). Between 1990 and 2022, woodland associated butterflies declined by 47% (as shown in Figure 5), although they have shown no significant change since 2017. Species in woodland displaying the greatest declines include the small copper (*Lycaena phlaeas*) and the gatekeeper (*Pyronia tithonus*), both of which favour woodland rides and clearings. In contrast, the ringlet (*Aphantopus hyperantus*), which favours similar habitats, has shown significant increases over the same period.

The Butterfly Conservation Trust also reports that the decline of the small pearl-bordered fritillary (*Boloria selene*) from South East England since the 1970s can be attributed to the reduction in management in woodlands, while the loss of such coppiced woodlands has been linked with declines of specific woodland species such as the high brown fritillary (*Fabriciana adippe*) (Butterfly Conservation, 2023). In contrast, natural range expansions in Scotland and conservation introductions in central England are encouraging for the chequered skipper (*Carterocephalus Palaemon*), while the purple emperor (*Apatura iris*) has also shown considerable range expansion in England (Butterfly Conservation, 2023).

In Scotland, butterflies are monitored through the Butterfly Monitoring Scheme (UKBMS) and the Wider Countryside Butterfly Survey. The all-species index increased by 35% and the generalist index increased by 47% between 1979 and 2022. The specialist index remained relatively stable. Woodland associated species such as the speckled wood (*Pararge aegeria*) and ringlet have shown significant increases since 1979. Pearl-bordered fritillary, another species preferring woodland clearings, has also shown significant increases following targeted conservation action (Nature Scot, 2023).

A recent study looking at invertebrate trends more broadly by investigating the relationship between increasing woodland cover and the long-term distribution trends of woodland-associated invertebrates found declines of the little emerald moth (*Jodis lactearia*), which favours open woodland (Bowler et al., 2023). The study also found a decline in species specialised to certain strata, namely shaded woodland floors and arboreal habitats, which may be linked to low heterogeneity in British woodlands.

When considering the suitability of woodlands for invertebrates, this heterogeneity or niche diversity is key and should be considered on a different scale. Spiders cannot easily be classified as woodland specialists but rather as microhabitat specialists, requiring specific niches which may be present in or outside woods. *Philodromus margaritatus* requires lichen-covered bark, for example, while veteran trees provide features needed by several species of conservation concern including the endangered *Midia midas*, vulnerable *Mastigusa macrophthalma* and near-threatened *Leviellus stroemi*. The endangered *Tuberta maerens* favours coppiced woods or glade-edge trees rather than closed-canopy woodlands, and many deep-litter *Linyphiidae* species need the shaded, cool, humid leaf litter provided within woodlands (Smith, 2024).

Discussion

One of the most important findings from the Bunce survey has been the general trend of continued stand development across the surveyed woods and the selective pressure this has exerted on woodland flora. Species capable of exploiting shaded and stable conditions have fared better than those requiring a more open canopy (see woodland condition and tree provenance sections for more information on the implications of continued stand development for long-term woodland regeneration and resilience). Species which thrive in more fertile conditions have increased in abundance while deer grazing, tree disease and a warming climate have also played a role in influencing the composition of woodland flora communities.

While the selective pressure exerted by the drivers outlined above benefits certain species, such as bluebell or holly, it negatively affects other species, such as wild strawberry or wood sorrel. It is also unlikely to be positive for the woodland landscape as a whole. The changes in species composition reported in the Bunce survey are generally consistent across sites, suggesting that the same species are recovering across Britain. This is likely to have important implications for woodland diversity at a landscape scale. Although some woodland communities naturally feature fewer species than others, it is generally accepted that a more diverse ecosystem is more resilient to threats and more able to adapt to change (Bullock et al., 2022). By keeping woodlands on their current trajectory which selects for a limited number of species, we risk reducing complexity, and therefore resilience (an emergent property of complexity), in British woodlands.

A system featuring high floral species richness will generally provide resources and niches for a wider range of associated species, and a structurally diverse woodland will provide niches for a greater variety of floral species to inhabit. The conditions described in the Bunce report are not conducive to this diverse state and may threaten the long-term conservation status of much of our woodland biodiversity. Woodland birds and butterflies are experiencing long-term declines across Britain, which can be linked to a decline in woodland condition and availability of specific niche spaces (JNCC, 2023b, 2023d, Butterfly Conservation, 2023). Dormice, which thrive in open early-successional woodland with mature trees (Wembridge et al., 2023), also continue to experience ongoing declines. It is important to note that drivers of species populations are complex, numerous and interacting, and factors in the wider landscape such as agricultural intensification and habitat fragmentation also contribute to species loss.

However, aiming to increase structural diversity of woodlands in order to increase complexity, while continuing to advocate for wider landscape scale solutions across a range of habitats, provides a practical action that can be taken within woodlands to reverse declining species trends.

There is continued debate around whether development of woodlands in Britain towards a climax community, whereby woodlands reach a stable and mature, high forest 'endpoint', is a good way to interpret woodland succession at all. The woodlands surveyed in the Bunce survey were themselves affected by timber removal prior to 1971, and it is not clear that the open conditions experienced in the first round of the survey are the natural state of things. However, woodlands in Britain have undoubtedly evolved to survive in dynamic and disturbed environments. Large herbivores previously native to Britain would knock down trees and create open spaces, and woodland associated species have adapted to this. Millennia of human activity will also have played a major role in shaping woodland species' capacity to respond to change. So how can this disturbance be built back into woodlands to increase structural complexity and habitat diversity? One option is to increase conservation management interventions within woodlands.

Structural complexity can be considered when creating new woodlands as part of the design phase, incorporating groves, open wooded habitats and glades (Woodland Trust, 2022). These woods will still, nonetheless, need time to establish to the point that they resemble mature woodlands and there will be a time lag between habitat creation and species colonisation (Fuentes-Montemayor et al., 2015, 2023a, 2023b). It is important that existing woodlands already present in the landscape are subject to increased restoration and ongoing active conservation intervention efforts, a proxy for the natural disturbance regimes that woodland species have evolved alongside, to maximise structural diversity and available niche space. Financial incentives are required which provide viable options for landowners to improve the condition of their woodlands through regular conservation management, in order to provide the niches required by woodland-associated biodiversity.

Initiating this shift towards widespread and concerted programmes of active management aimed at facilitating nature recovery at landscape scale is even more important in the face of climate change. Previous studies have demonstrated that broadleaved woodlands are second only to calcareous grasslands in terms of their geographical exposure to climate change in Britain (Wilson and Pescott, 2023), and that the effects of climate change in broadleaved woodlands may be lagged (Bertrand et al., 2011) such that the woodlands we see today are yet to have fully responded to historical change.

Managing woods in a way that maximises diversity and complexity will help to mitigate the anticipated effects of climate change by enhancing resilience.

Increasing woodland management may bring with it challenges, however. Gaps and rides created by management may benefit a range of species which favour open glades but will be exposed to greater climatic variation away from the moderating effect of the canopy. This will be an important factor shaping the appropriate management at a given woodland, and the interactions between gap creation and other drivers of biodiversity change such as eutrophication and herbivory will need to be monitored closely. For example, if deer numbers remain high it is possible that tree regeneration may be suppressed in woodland gaps, leading to the development of relatively open communities comprising grazing-tolerant plants. These communities, increasingly shaped by herbivory, may also create the conditions necessary for shade-intolerant plant species to colonise woodland more widely.

Although there is a need for diverse woodland structure featuring open spaces and young regenerating trees, old growth closed canopy woodland provides many features for biodiversity, from nesting features for birds and large pieces of fallen or standing deadwood, to shaded and stable conditions suitable for specialist ground flora, fungi and invertebrates. Recovering bat populations in Britain are likely to have benefited from an increase in high forest (Burns et al., 2016), which emphasises the need to carefully consider which management interventions are appropriate on a site-by-site basis. Finding the balance between maximising biodiversity benefits and limiting exposure to climatic extremes will be a challenge in woodland conservation in the coming decades, and unlocking resources for thorough site-based assessments when undertaking management should be a priority in the short term to provide the most benefit for woodland biodiversity across Britain.

Conclusion

The results from the Bunce survey and indicator data for a wide range of woodland taxa indicate that much of our woodland wildlife is continuing to decline. Many of the woodland-associated species experiencing ongoing declines are those which require open spaces and diverse vegetation structure within woodlands to thrive, and this trend is consistent with the general trend of canopy closure described in the Bunce report. Additional drivers affecting the composition of woodland flora include eutrophication, tree disease and deer browsing, while a warming climate is also favouring certain species. We urgently need our woodlands to be resilient to the threat of climate change and extreme weather events. Increasing management, and therefore niche availability within woodlands is one way to build back complexity, which will have the dual benefit of increasing resilience while providing space and resources for a wider range of woodland species.

Evidence gaps and barriers

- Continued development of funding for conservation interventions, management plans and site assessment is required.
- Active woodland management should be seen as being as important a factor for woodland biodiversity as woodland creation.
- Exploration of the viability of markets for products which create more

diverse woodland structure (coppicing, charcoal production etc) would be beneficial here. Understanding the mechanisms capable of influencing or creating markets for these products on a local scale and the potential to reinstate large-scale active management operations in woodlands will be key.

- Specific grants for woodland grazing should also be explored, and landowners should be aware that some grazing is likely to be more beneficial than no grazing at all.

It is also worth noting that while the main thread of the discussion is pushing for increased management in the form of conservation interventions, this is context specific, and it is important to remember that not all species will benefit from increased dynamism. Finding balance between management and old growth woodland will be key. Finding balance between gap creation and other drivers of change (browsing, eutrophication etc) will also be essential in managing healthy and diverse woodlands on a landscape scale.

Lastly, much of the population declines outlined here do not account for historical woodland cover. Although woodland cover is increasing in the UK, it is possible populations continue to decline as a result of historical woodland loss, lagged effects and extinction debts. Further study exploring to what level population declines are driven by extent and condition of woodland would be useful here. However, as woodland specialists and species of open conditions appear to be particularly struggling, it seems that woodland condition is an important factor and we are justified in focusing on advocating for increased conservation management interventions.

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Carbon

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Abstract

- The carbon cycle is dynamic and complicated. Terrestrial ecosystems take up and release vast amounts of carbon through the processes of photosynthesis, respiration and decomposition.
- Woodland ecosystems tend to be carbon sinks whereby the amount of carbon being sequestered from the atmosphere is greater than the amount of carbon being released via respiration and decomposition. This results in woodland carbon stocks increasing over long periods of time (c. 100 years).
- The majority of terrestrial carbon is stored in the soil - there is nearly twice as much carbon stored in the soil as is stored in all global terrestrial ecosystems (i.e. all the plants, trees and animals) and the atmosphere combined.
- Woods and trees play an important role in the UK's climate change mitigation and net zero strategy, however we are significantly off track in meeting the required woodland creation targets to achieve the net zero pathway set by the Climate Change Committee (CCC).
- The total carbon stock of UK forests is about one billion tonnes of carbon (1095 Mt C).
- England 400 Mt C, Wales 92 Mt C, Scotland 559 Mt C, Northern Ireland 45 Mt C
- The annual rate of net carbon dioxide removal by UK forests was estimated to be 18.3 Mt CO_2e which represents around 4.7% of the UK's total annual territorial emissions (2023)
- Under the CCC's balanced net zero pathway scenario, it is estimated that meeting woodland creation targets of 30,000 hectares of woodland per year, alongside improved management of existing woodlands and expansion of hedges and agroforestry could result in an annual carbon sequestration of ~29 Mt CO_2e by 2050.
- Woodland creation on carbon-rich soils may limit net carbon sequestration or even lead to a net loss of carbon over decadal timescales.
- The impacts of woodland creation on mineral soils are less likely to result in the loss of soil carbon.
- Carbon sequestration into tree biomass is highly correlated with growth rates of trees. As a result, fast-growing conifers sequester more carbon over short timescales compared to slower-growing deciduous trees. However, over longer timescales (c. 100 years) the difference is smaller due to carbon losses associated with harvesting and restocking commercial conifer stands.
- Carbon models of different woodland creation scenarios are useful in providing indicative assessments of predicted carbon sequestration rates. However, real-world site conditions and balancing multiple objectives for woodland creation means that scenarios are not interchangeable between

sites. The right tree in the right place for the right reasons must drive decision-making so that we balance outcomes for nature, climate and people.

- To contribute to climate change mitigation and net zero targets, the timing of woodland creation matters. This is due to the slow initial rates of carbon sequestration as trees grow. It's estimated that missed creation targets between 2020-2021 and 2023-2024 would have removed 8.5 million tonnes of carbon dioxide (Mt CO_2e) by 2050.
- It's not just new woodland creation that can remove atmospheric carbon. The carbon stock of living trees in ancient and longstanding woodlands is predicted to double over the next 100 years.
- Empirical studies have shown that ancient and longstanding woodlands can remain a net carbon sink even as they age and growth rates slow, with carbon sequestration exceeding losses from decomposition and respiration. It is suggested that historical woodland management may be a key factor in determining the current carbon balance of ancient woodlands.
- New assessments of above ground carbon stocks using terrestrial LiDAR demonstrate that traditional methods may be underestimating the carbon stock of the above ground biomass of semi-natural native broadleaf woodland by nearly 80%. With 50% of the carbon stored in above ground biomass being found in less than 7% of the largest trees.

Introduction

Woods and trees are a vital part of the UK's Net Zero Strategy for climate change mitigation (Committee on Climate Change, 2020b) by removing and storing carbon dioxide (CO_2) from the atmosphere. The Climate Change Act 2008 commits the UK government to meet net zero carbon emissions by 2050; however, there are significant risks to achieving the level of emissions reductions and carbon removals needed to meet these commitments. Alongside vast reductions in fossil fuel-derived greenhouse gas (GHG) emissions from industry, transport and energy generation, we must greatly reduce GHG emissions from land use whilst also increasing atmospheric carbon removals through land management, habitat creation and restoration. It is vital that we also recognise that carbon dioxide removals from woods and trees only have the capacity to offset unavoidable residual GHG emissions following effective decarbonisation and emissions reductions measures from other sectors.

A recent UK CCC progress report to the UK Parliament concluded that although some progress has been made to reduce emissions, only a third of the emissions reductions required to achieve the interim 2030 target are covered by credible plans. They warn that the agriculture and land use sectors have made very little progress with almost all indicators for the roll-out of nature-based solutions and carbon removals being off track (CCC, 2024).

> **“Tree planting and peatland restoration rates are significantly off track and will both need to more than double to get as close as possible to the UK’s targets of 30,000ha new woodland creation per year by 2025 and 32,000ha peatland restoration per year by 2026” (CCC, 2024)**

Alongside habitat creation, protecting and restoring existing carbon stores in semi-natural habitats such as ancient and longstanding woodlands, species-rich grasslands and peatlands will be essential to achieving the UK’s net zero commitments.

The global context for understanding carbon dynamics between ecosystems and the atmosphere

Woods and trees play an important role in global carbon cycling. Like all green plants, trees use the process of photosynthesis to convert carbon dioxide from the atmosphere, water and energy from sunlight into sugar molecules and oxygen. These sugars are then used to produce the building blocks for plant growth and repair. Photosynthesising organisms (plants, algae and bacteria), and the chemical energy that they produce, are the foundation of most of the life on Earth.

Globally, the carbon that is sequestered annually by photosynthesis (called gross primary production or GPP) is estimated to be ~120 Pg C y^{-1} (90 -130 Pg C y^{-1}) (Lal, 2008) , which is about 12 times greater than current estimates for global fossil fuel CO^2 emissions 10.0 Gt C yr^{-1} (Friedlingstein et al., 2022). However, it is vital to recognise the difference between carbon sequestration and storage. Most of the carbon that is sequestered from the atmosphere via photosynthesis into plant biomass is in turn replaced by carbon that is released back into the atmosphere via plant respiration (40 to 60 Pg y^{-1}) and organic matter decomposition (40 to 68 Pg C y^{-1}). The significance of this is that the exchange of carbon between the atmosphere, biosphere and the soil is near equilibrium, with carbon sinks developing slowly over hundreds, thousands or millions of years due to small changes between carbon uptake (photosynthesis) and release (respiration). The introduction of carbon from fossil sources, which had previously been stable over geological time, has altered this balance and increased the concentration of CO^2 in the atmosphere to 419.3 parts per million (ppm) (51% above pre-industrial levels) in 2023, resulting in climate warming (Friedlingstein et al., 2022; Lan, Tans and Thoning, 2023). The capacity of natural carbon sinks, including semi-natural ecosystems, to take up this additional carbon is variable and uncertain. A global estimate between 1990 and 2007 reported that forests were a net sink of 1.1±0.8 Pg C y^{-1} (Pan et al., 2011).

To reduce the most extreme impacts of climate change caused by GHGs that have already been emitted from fossil fuels and land use change, we must stop emitting additional GHGs and remove as much carbon as possible from the atmosphere. One way of achieving the latter is by massively expanding the land cover of semi-natural habitats including woods and trees and increasing the proportion of the active carbon pool that is sequestered into living biomass or the soil.

Methods

A non-exhaustive literature review was conducted on the role of UK woods and trees in carbon dynamics using Google Scholar to find published peer-reviewed literature as well as grey literature, websites and key datasets. Official UK Government statistics and reporting from the National Forest Inventory were also included. Searches focused on UK studies of native woodlands with national datasets including all forest cover being used for national estimates of carbon stocks and sequestration rates. Detailed comparisons between native woodlands and non-native plantation forestry were out of scope of this report. Primary search string used was 'forest*' OR 'wood*' OR 'tree* AND 'carbon' OR 'carbon dioxide' OR 'carbon balance' OR 'carbon stock' OR 'net ecosystem exchange' OR 'net ecosystem productivity*' Studies of UK woodlands were prioritised. Reference lists were checked to see if they contained additional relevant sources.

Carbon models presented in Figure 1 and Figure 2 were generated using the Woodland Carbon Code (WCC) Carbon Calculation Spreadsheet version 2.4.1 April 2024. Species mix, spacing and yield class values were derived from alignment with the Woodland Trust's *Woodland Creation Guide* (Herbert et al., 2022), WCC lookup tables and, where possible, cross-checked against the Forest Research Ecological Site Classification (ESC) web-based decision support tool to provide realistic creation scenarios for a range of sites. Results were plotted using 'ggplot2'.

Results/Discussion

The role of UK woods and trees in meeting net zero and climate change mitigation targets

In order to assess the role of UK woods and trees in meeting net zero and climate mitigation targets, it is important that we consider both the size and stability of existing woodland carbon stocks, as well as the potential for existing woodland cover and additional woodland creation to remove carbon dioxide from the atmosphere. Creation, restoration and protection objectives will each play a vital role in meeting the UK's climate change mitigation targets.

UK woodland carbon stock

As of March 2024, the area of woodland cover in the UK is estimated to be 3.28 million hectares, which is ~13% of the total land area; 19% in Scotland, 15% in Wales, 10% in England, and 9% in Northern Ireland (Forest Research, 2024). The total carbon stock of UK forests is about one billion tonnes of carbon (1095 Mt C; Table 1). Tree biomass (above and below ground) accounts for approximately 23% of the total woodland carbon stock compared to 70% which is stored in forest soils (Forest Research, 2023b). Soils store a vast amount of carbon over long time periods and are the third largest global carbon pool. Indeed, there is nearly twice as much carbon stored in the soil as is stored in all global terrestrial ecosystems (i.e. all the plants, trees and animals) and the atmosphere combined (Lal, 2008).

Table 1. Forest carbon stock by country. Million tonnes of carbon (Mt C). Data reproduced from Forest Research submission to FAO Global Forest Resources Assessment (FRA) 2020. Details on methods used to produce these estimates are outlined in (Forest Research, 2023b). Note, data has been converted from million tonnes carbon dioxide equivalent ($MtCO_2e$) to million tonnes carbon (Mt C).

Carbon Stock	England	Wales	Scotland	Northern Ireland	UK
Above ground biomass	92	17	71	4	184
Below ground biomass	33	6	25	1	66
Deadwood	17	4	19	1	41
Litter	22	5	23	2	52
Soil (1m depth)	236	59	421	37	753
Total forest carbon	**400**	**92**	**559**	**45**	**1095**

While the soil carbon stock of UK woodlands is vast, it is important to caveat that the estimates for forest soils includes forest cover that has previously been planted on deep peat. It is estimated that around 76% of this forest cover on peat comes from historic afforestation of organic and peaty soils in Scotland (Evans et al., 2017). This skews the data for the total UK carbon stock and represents a significant risk for current and future emissions if these peatland sites are not rewetted and restored. Evans et al 2017 estimated that around 16% of UK peat area has forestry cover, predominantly drained conifer plantations, and could be contributing 4.6 Mt CO_2e yr^{-1} or 20% of all GHG emissions from degraded UK peatlands.

UK-wide assessments of woodland soil carbon are challenging, as carbon stocks vary significantly between soil types and surveys have sampled soil to different depths. The most extensive assessment of UK forest soils was the BioSoil project (Vanguelova et al., 2013) which evaluated carbon stocks in forest soils covering 167 plots across Great Britain. This survey covered a range of soil types under broadleaf and conifer stands and took five replicate samples at five incremental depths ranging from 0-80cm. This data was then extrapolated down to 100cm depth to allow for comparison with other datasets on forest soil carbon. The average carbon stock for the seven main soil types down to an 80cm depth ranged between 108 t C ha-1 (rankers and rendzinas) and 448 t C ha^{-1} (deep peats) (Table 2). This data demonstrates the large variability of woodland soil carbon stocks under different soil types as well as the greater variability of soil carbon stocks found in the most carbon-rich, organic peaty gleys/podzols and deep peats (Vanguelova et al., 2013). This is particularly important to note as these carbon-rich soils represent a significant proportion of both current woodland cover and available land for woodland creation, especially in UK upland sites and Scotland (Rees et al., 2018).

Recent studies have suggested the establishment of native woodland on these organic soils may result in soil carbon loss (Matthews et al., 2020)

(Warner et al., 2022), which may not be recouped by gains in carbon stored in tree biomass over decadal timescales (Friggens et al., 2020). These studies suggest that tree planting may not result in a net gain in ecosystem carbon storage and may even decrease net ecosystem carbon stocks within the timeframes required to tackle the climate crisis and to meet net zero targets. However, this is an active area of research and there are some caveats and evidence gaps that could strengthen our understanding and help contextualise these emerging results.

Commonly used space-for-time studies rely on matching comparable woodland establishment and control (non-wooded) plots and can lack baseline data of soil properties. While this allows us to explore carbon dynamics over longer timescales, uncertainties arise due to differences in the initial soil conditions, the potential impact of variable tree survival rates, or the preferential establishment of woodland based on initial conditions that may correlate with lower soil carbon contents, for example drier soils. It is also important that future studies explore carbon stocks in deeper mineral horizons and include litter and fermentation layers in assessments of net ecosystem carbon stocks. We should also remain cautious about making land use decisions based solely on carbon dynamics and ensure that short-term objectives to meet net zero targets via woodland creation are balanced against long-term carbon impacts (>100 years) alongside nature recovery and ecosystem service objectives.

Impacts of woodland creation on the carbon stocks of mineral soil are less likely to result in the loss of soil carbon and have been reported to lead to the accumulation of soil carbon over long time periods (Poulton et al., 2003; Benham, Vanguelova and Pitman, 2012; Ashwood et al., 2019). Increases in soil carbon may be greatest on former arable sites with mineral soils where previous site management has reduced soil carbon stocks. The UK National Greenhouse Gas Inventory estimates that converting arable land to forestry land increases the soil carbon by 31 t C ha^{-1} (Brown et al., 2020). Morison et al., 2012 highlighted that, although there is a lack of empirical data, afforestation of mineral soils is expected to increase soil carbon at typical rates of between 0.14 to 0.46 t C ha^{-1} yr^{-1}. Field-based assessments of soil carbon following tree planting have found similar rates of soil carbon accumulation. Falloon et al., 2004 estimated a mean increase in soil carbon of 0.53 t C ha^{-1} yr^{-1} from newly planted shelterbelts and hedgerows on arable land over a 40-year period, while Upson and Burgess, 2013 reported a 0.46 t C ha^{-1} yr^{-1} increase in soil carbon following conversion from arable to silvopastural system. In contrast, some studies have reported that moving from a grassland to forested/ silvopastural system can lead to a small loss of soil carbon (although isolated to the 0-10 cm depth) or lead to no net change in soil carbon stock (Beckert et al., 2016; Fornara et al., 2018; Drexler, Gensior and Don, 2021).

It is important that the uncertainties associated with both carbon stocks and carbon sequestration rates into woodland soils are reported and included in carbon accounting models. We should take a precautionary approach both when changing land use and when reporting soil carbon gains in carbon accounting methodologies. As highlighted by Vanguelova et al., 2013 (Table 2), the ranges of soil types, geology and organic matter composition of our woodlands can all play a significant role in how soils interact with woods and trees. Studies exploring the rate of soil carbon accumulation also demonstrate the long timescales required for carbon to be sequestered into the soil. If we assume a typical annual rate of soil carbon accumulation of 0.3 t C ha^{-1} yr^{-1} (0.14 - 0.46 t C ha^{-1} yr^{-1}; Morison et al., 2012) following woodland establishment on mineral soil, and the UK National Greenhouse Gas Inventory estimates that converting arable (mineral soil) to forestry land increases the soil carbon by 31 t C ha^{-1} (Brown et al., 2020), we can estimate that it will take approximately 100 years for mineral soils to develop soil carbon stocks equivalent to woodlands on mineral soils (Vanguelova et al., 2013 ; Table 2). This assumes that sequestration rates are linear over time and does not consider potential changes to sequestration rates resulting from the impacts of climate change. This highlights how long it takes for soil carbon stocks to develop and the importance of protecting soils from degradation and damaging land use change.

Table 2. Mean woodland soil carbon stocks (t C ha^{-1}) between 0-80 cm depth across different soil types. Reproduced from the BioSoil project (Vanguelova et al., 2013)

Soil Type	Mean carbon stock 0-80cm (t C ha-1)	Standard errors of the mean
Rankers and rendzinas	108	24
Brown earths	135	6
Podzols and ironpans	136	16
Surface-water gleys	147	10
Groundwater gleys	155	18
Peaty gleys/podzols	321	40
Deep peats	448	36

The contributions of the UK's woods and trees to carbon sequestration

Forestry is the largest net sink within the UK's land use, land use change and forestry sector (LULUCF), with approximately equal contributions from broadleaf and conifer woodlands. The annual rate of net carbon dioxide removal by UK forests was estimated to be 18.3 Mt CO_2e (Brown et al., 2020; Forest Research, 2023b) which represents around 4.7% of the UKs total territorial greenhouse gas emissions[1]— estimated to have been 384.2 MtCO_2e in 2023 (Department for Energy Security and Net Zero, 2024) . The UK's Committee on Climate Change (CCC) recommends increasing UK forestry cover from 13% to at least 17% by 2050 by planting around 30,000 hectares of woodland each year. Under the CCC's balanced net zero pathway scenario, it is estimated that meeting these creation targets alongside improved management of existing woodlands and expansion of hedges and agroforestry could result in an annual carbon sequestration of ~29 Mt CO_2e by 2050 (Thomson et al., 2020) ; approximately 7.5% of the UK's current annual territorial emissions. In contrast, the CCC 'business as usual' (BAU) scenario has modelled annual sequestration of the UK's forests to fall to

~11 Mt CO_2e by 2040-2050 due to the changing age composition of UK forests and felling restocking cycles (Thomson et al., 2020). These analyses excluded the additional benefit of timber production substituting building materials and energy production from, which could be as much as 14 Mt CO_2e (Committee on Climate Change, 2020a). However, the majority of the UK's timber production currently goes into short life cycle products such as pallets and fencing (Forest Research, 2023a) and we will need to enhance the quality of timber production to meet the more ambitious substitution emission reductions through supplying high-quality timber for construction.

A recent Forest Research report Quantifying the Sustainable Forestry Carbon Cycle by (Matthews et al., 2022) compared rates of carbon sequestration from different types of woodland in the UK. The analysis includes 12 woodland types covering coniferous, broadleaf and natural regeneration of native woodland as well as a range of management levels. They assessed these woodland creation scenarios over a range of time scales between 2022-2100 using the CARBINE model. The results of this study showed that all modelled woodland creation scenarios delivered carbon sequestration over the 2022 – 2100 period. Over short time periods (2022 – 2050), fast-growing coniferous plantation resulted in the highest uptake of CO^2, with lightly managed broadleaf and naturally recolonised woodland sequestering substantially less carbon. The carbon sequestration rate in broadleaf woodland was 0.9 to 1.6 tCO_2 ha^{-1} yr^{-1}, while the rate in coniferous woodland was 1.8 to 12.0 tCO^2 ha^{-1} yr^{-1}. Over longer time periods (2022-2100), there was a smaller difference between broadleaf and coniferous woodland types, ranging from 3.9 to 5.5 tCO^2 ha^{-1} yr^{-1} and 3.6 to 10.6 tCO_2 ha^{-1} yr^{-1}, respectively.

The rate of CO_2 sequestration in carbon accounting models such as CARBINE and WCC is strongly correlated with the assumed growth rate of the trees from yield class models. Assessments typically model broadleaf woodland types with a yield class of 2 - 6 and productive conifers ranging from yield class 8 up to yield class 24 for fast growing Sitka spruce. This is likely to vary significantly with site conditions and over time as weather and climate change. It is also likely that the upper estimates for productive conifer plantation (yield class 24) are ambitious and likely only achievable in optimal conditions and with intensive management interventions and ground preparation. It is important that this uncertainty is considered when trying to apply these results at a regional or UK-wide scale. Whilst comparisons between woodland types are informative and highlight the role of different creation scenarios, it is important that the UK balances multiple outcomes from woodland creation when making land use decisions. Woodland creation scenarios used in these types of analyses are not always interchangeable between different sites and locations in the UK. In defining the parameters of the model, assumptions were made about factors such as weather, soil type, ground vegetation and species mix. This is done to ensure that model parameters are suitable for the creation objectives and site conditions.
As such, decisions about which woodland type is suitable on a given site should not be made solely on model output in an attempt to maximise carbon sequestration. These decisions should be made by ecological and environmental impact assessments regarding site suitability and species choice and by balancing aims and outcomes for carbon sequestration,

biodiversity and conservation, timber production and the provision of ecosystem services. Nature recovery, timber production, carbon sequestration and ecosystem service objectives will all need to be delivered to mitigate and adapt to the impacts of climate change.

Using the Woodland Carbon Code (WCC) calculator, we can further explore the carbon storage delivered by different woodland creation scenarios that prioritise nature recovery and habitat creation objectives. Figure 1 shows cumulative carbon sequestration ($CO_2e\ ha^{-1}$) over 100 years for three woodland creation scenarios that represent typical conservation-focused woodland designs in which species mix, spacing and management are aligned with the Woodland Trust's Woodland Creation Guide (Herbert et al., 2022). Yield class assumptions are derived from WCC lookup tables and, where possible, cross-checked against Forest Research's Ecological Site Classification tool (ESC) to provide realistic creation scenarios for a range of sites. These models demonstrate the significant impact of site conditions, soil type and species choice for determining net carbon storage in above-ground biomass after 100 years. It's also important to reflect that each of these creation scenarios are predicted to result in net carbon sequestration over long time periods while balancing carbon, biodiversity and ecosystem service objectives. This is most clear for uplands sites with carbon rich, organo-mineral soils in which multiple conservation priorities need to be considered alongside protecting existing soil carbon stocks from ground preparation or fast growing and dense woodland creation.

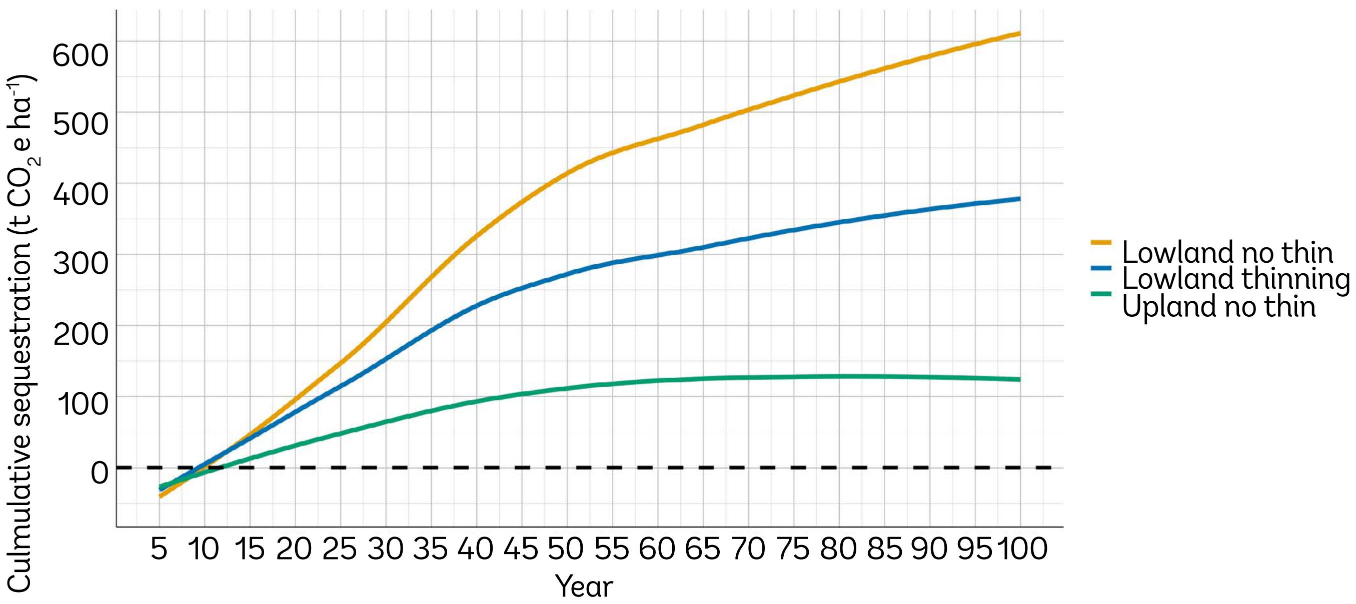

Figure 1. Cumulative carbon sequestration of three woodland creation scenarios over 100 years ($tCO_2\ ha^{-1}$), modelled using the WCC Carbon Calculation Spreadsheet version 2.4.1 April 2024. Species mix, spacing and yield class values were derived from alignment with the Woodland Trust's Woodland Creation Guide, WCC lookup tables and cross-checked against ESC where possible to provide realistic creation scenarios for a range of sites. The 'lowland' scenarios modelled a broadleaf mix planted on mineral ex arable soil, while the 'upland' scenario modelled lower density planting of a broadleaf mix on organo-mineral soil.

The rate of carbon sequestration for woodlands is not linear and varies with species and site conditions. Initially, as trees grow, carbon sequestration rates tend to be relatively slow before peaking at around ~30yrs post-establishment. After this period of rapid growth, carbon sequestration rates slow down and net sequestration approaches equilibrium with respiration from trees and decomposition processes. Woodlands in this later phase can remain a net sink but this is variable and dependant on levels of management and disturbance (see below). However, they are highly significant long-term carbon stores. This model of woodland growth highlights the importance of when trees are established to determine their contribution to atmospheric carbon removals by 2050 net zero targets.

The 'lowland thinned scenario from Figure 1 is estimated to sequester and store ~377 t CO_2e ha^{-1} over 100 years. If we adjust the planting date (2015, 2025 and 2035), we can model how much CO_2 will be sequestered by 2050 and therefore contribute to the UK's net zero commitments. We could also think of this as the amount of carbon foregone if we fail to meet woodland creation targets. If we established the 'lowland thinned' woodland in 2015, we would have sequestered 216 t CO_2e ha^{-1} by 2050 net zero targets compared to just 16 t CO_2e ha^{-1} if we plant in 2035. The percentage of total predicted carbon sequestration and storage this planting scenario could deliver by 2050 given different establishment dates are as follows: 2015 = 57.2%, 2025 = 22.8%, 2035 = 4.4%. In other words, the amount of CO_2 that could have been sequestered per hectare by 2050 if we planted in 2015 rather than 2035 is approximately equivalent to the lifetime emissions of five medium-sized cars[2]. If we scale this up to the UK Government's target of 30,000 hectares per year of new woodland creation, we start to get a sense of the huge role that woods and trees can play in removing CO_2 from the atmosphere and the importance of meeting woodland creation targets.

Analysis by Carbon Brief exploring the impact of the UK failing to achieve its woodland creation targets, calculated that missed tree planting targets between 2020-2021 and 2023-2024 would have removed approximately 8.5 million tonnes of carbon dioxide (Mt CO_2e) by 2050 (Gabbatiss and Viisainen, 2024). These analyses highlight the importance of woodland creation targets being met as soon as possible if they are to provide a significant contribution to carbon removals by net zero targets. If we miss these creation targets, it will require significantly more land and woodland creation to deliver the same amount of carbon removals by 2050. We must also emphasize that although the contribution of woodland creation closer to 2050 will have a reduced impact on net emissions by net zero target dates, they will still deliver carbon sequestration and storage over longer timescales whilst also delivering multiple climate adaptation, ecosystem service and nature recovery objectives.

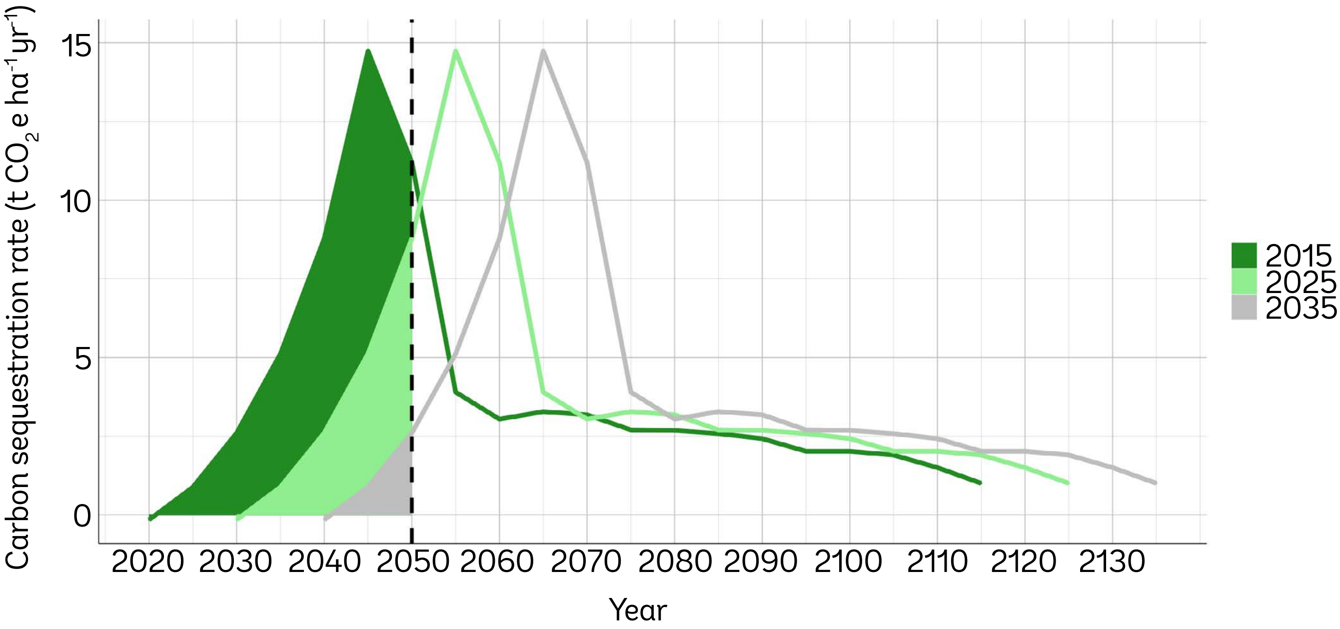

Figure 2. Carbon sequestration rate (tCO_2 ha^{-1}) of a typical lowland mixed broadleaf woodland creation scheme on mineral soil over 100 years with thinning management. Modelled using the WCC [Carbon Calculation Spreadsheet version 2.4.1 April 2024]. Species mix, spacing and yield class values were derived from alignment with the Woodland Trust's Woodland Creation Guide, WCC lookup tables and, where possible, cross-checked against ESC to provide realistic creation scenarios for a range of sites.

The contribution of new woodland creation to remove atmospheric carbon is a vital part meeting net zero targets; however, less focus has been given to the important role of our existing woodland habitats. As we reported in the first State of the UK's Woods and Trees report in 2021, the total amount of carbon (in living trees) in ancient and long-established woodlands across Great Britain is estimated to be 77 million tonnes, which is approximately 36% of all woodland carbon in living trees within Great Britain (Reid et al., 2021). Analysis by Forest Research using the National Forest Inventory (NFI), modelled that the carbon stock of living trees within ancient and long-established woodland is set to double over the next 100 years from c. 77 million tonnes to c. 155 million tonnes (Reid et al., 2021; Figure 3). These models are based on a no-harvest scenario and reflect the amount of ancient woodland containing younger trees that still have strong growth and sequestration ahead of them. While these assumptions are likely realistic for ancient woodlands with predominantly native species, there is greater uncertainty regarding degraded ancient woodlands or plantations on ancient woodland sites (PAWS) which may undergo more significant felling to transition from plantation forestry to native species.

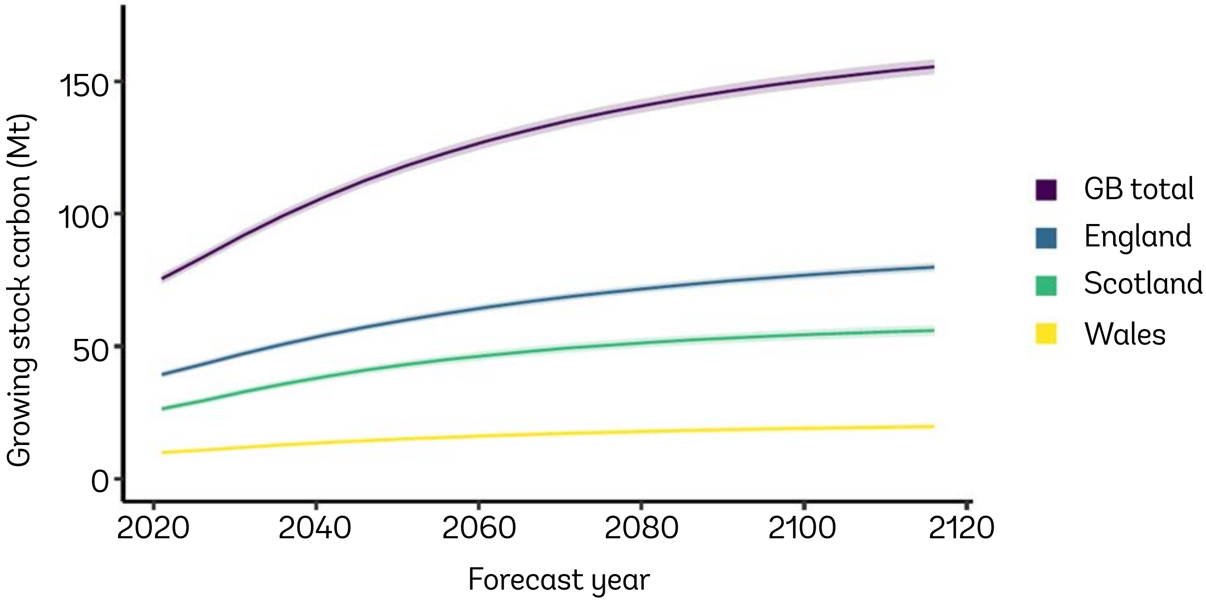

Figure 3. The 100-year forecasts for average annual growing stock volume in ancient and long-established woodlands in Great Britain, England, Scotland and Wales (Forest Research NFI data reported by (Reid et al., 2021).

While these analyses demonstrate the huge role the UK's woods and trees play in tackling the climate crisis and meeting our net zero commitments, it is important that we acknowledge that meeting even the most ambitious afforestation targets will still require significant decarbonisation across all UK sectors. Collectively we must pursue wider societal and political action to reduce global greenhouse gas emissions and increase the protection and restoration of our ecosystems. We must be powerful advocates for the need to live within planetary boundaries and rebuild our relationship with the natural world.

Are ancient woodlands and semi-natural native woodland a carbon sink?

The role of ancient or 'old growth' forest in the carbon cycle has been long debated in the scientific literature (Luyssaert et al., 2008, 2021; Gundersen et al., 2021). This work has largely focused on modelling the net ecosystem exchange of carbon and whether old forests are net carbon sinks, sources or in equilibrium (i.e. when the sequestration of carbon through photosynthesis is balanced by carbon being released via respiration and decomposition). These studies demonstrate that 'old growth' forests continue to act as a carbon sink of between 1.6 ± 0.6 t C ha^{-1} yr^{-1} (Gundersen et al., 2021 re-analysis of Luyssaert et al., 2008) and 2.4 ± 0.8 t C ha^{-1} yr^{-1} (Luyssaert et al., 2021). These analyses highlight the challenge of generalising these processes due to uncertainty around forest management history, growth rates and measuring ecosystem level GHG fluxes. What is clear is that mature, 'old growth', and ancient woodlands contain vitally important carbon stores and that their protection is essential to avoid significant losses of carbon into the atmosphere. As woodlands mature, it is likely that rates of net carbon sequestration plateau off as carbon assimilation into biomass comes into equilibrium with losses from decomposition and autotrophic respiration. However, at an individual tree level, it has been shown that large mature trees continue to accumulate carbon and play a disproportionate role in forest carbon storage relative to their numbers in the wider tree population (Stephenson et al., 2014).

It is also worth noting that a lot of the international literature on this topic refers to 'old growth' forests which have been defined in multiple ways. The original term was developed by the logging industry in western United States and Canada during the 1900s and later became a significant term used by conservationists to campaign for the protection of late-successional stands of high-volume, quality timber prized by the forestry industry (Alexander et al., 2002). For the purposes of this report, we will focus on modelled and measured assessments of UK ancient and long-established woodlands to avoid comparisons with woodlands not representative of the UK.

In order to explore this topic, we need to define a few key terms. 1) gross primary productivity (GPP) is the amount of carbon dioxide that is assimilated via photosynthesis in to organic compounds, 2) ecosystem respiration (Reco) is the total amount of organic carbon that is released from an ecosystem via autotropic (plant) and heterotrophic (decomposition by bacteria and fungi) respiration, 3) net ecosystem productivity (NEP) is the difference between the total amount of carbon fixed via photosynthesis (GPP) and the total amount of carbon released via ecosystem respiration; therefore, NEP describes the total accumulation of carbon by an ecosystem. An ecosystem is a carbon sink when the rate of GPP exceeds the rate of ecosystem respiration.

As we discussed in the section above, the first *State of the UK's Woods and Trees* reported that the carbon stock of living trees within ancient and long-established woodlands is set to double over the next 100 years from c. 77 million tonnes to c. 155 million tonnes (Reid et al., 2021; Figure 3). While this provides vital insight into the age structure and potential future growth rates of these woodlands, it only tells part of the story of woodland carbon cycling. For these woodlands to be a net carbon sink, sequestration into living biomass must exceed losses by autotropic and heterotrophic respiration. There are only a few studies that we are aware of that have directly measured net ecosystem carbon exchange in UK ancient and semi-natural native broadleaf woodland (Table 3). These studies require the use of either an eddy covariance system or field-based measurements as a proxy for net ecosystem exchange using infrared gas analysers (IRGA) to measure gas fluxes alongside tree growth and decomposition rates (sometimes referred to as 'biometric' in the literature). Both approaches, but especially eddy covariance and flux towers, are expensive and require calibration. Most of the studies that measured the net exchange of carbon at a woodland level are either outside the UK or based on productive conifer forests; an overview of these wider studies can be found in Wilkinson et al., 2012.

Net ecosystem productivity (NEP) of UK native woodland ranges from 1.2 t C ha^{-1} yr^{-1} in an ancient semi-natural woodland (Thomas et al., 2011) up to 4.8 t C ha^{-1} yr^{-1} for a productive oak plantation (>90 years old) (Wilkinson et al., 2012). Gross primary productivity (photosynthesis) was similar between different native broadleaf woodland; however, ecosystem respiration was higher in the ancient semi-natural woodland which drove the differences in net ecosystem productivity. This is likely due to the high proportion of deadwood and decomposing material found in ancient woodlands (Patenaude et al., 2003; Butt et al., 2009). These studies demonstrate that ancient semi-natural woodlands can continue to act as a net carbon sink even as they mature and decay and decomposition rates increase. It is hard to disentangle whether reported net sequestration values are intrinsic to UK ancient and semi-natural woodlands or if historical management or weather patterns are dominate controls for net sequestration. Disturbance, coppicing and thinning management within a woodland may increase GPP by promoting tree growth, while the removal of brash and the extraction of mature trees may decrease deadwood and decomposition rates (ecosystem respiration). Wilkinson et al., 2016 reported that thinning management had no discernible impact on carbon balance, suggesting that annual variation and increased photosynthetic rates in ground vegetation due to canopy gaps made up for lower photosynthesis by the canopy. Interannual weather patterns, phenology and extreme weather events such as droughts, may all lead to forest carbon sink shifting to being a net carbon source (Valentini et al., 2000; Ciais et al., 2005; Powell et al., 2006; Pereira et al., 2007; Noormets et al., 2008). The duration and size of these shifts towards being a carbon source are highly uncertain and require further study. While the impacts of climate change alongside other threats such as pests and diseases on the net carbon balance of UK woodlands are hard to predict, it is evident that they play a hugely important role in carbon storage and are currently significant carbon sinks.

Table 3. field measurements of carbon sequestration and respiration from UK broadleaf woodlands. Units are expressed as t C ha^{-1} yr^{-1}; NEP = net ecosystem productivity, GPP = gross primary productivity, Reco= ecosystem respiration. * Indicates carbon fluxes were estimated using field based biometric sampling of key ecosystem productivity and respiration components.

Woodland type	NEP	GPP	Reco	method	Location	Reference
Oak plantation >90 yrs	4.86	20.34	15.48	eddy covariance	Alice Holt Forest, Hampshire, England	(Wilkinson et al., 2012)
Ancient semi-natural woodland >200yrs	1.2	21.1	19.8	eddy covariance	Wytham Woods, Oxfordshire, England	(Thomas et al., 2011)
Ancient semi-natural woodland >200yrs	1.7*	22*	20.3*	Biometric	Wytham Woods, Oxfordshire, England	(Fenn et al., 2015)

Woodland type	NEP	GPP	Reco	method	Location	Reference
Oak dominated 118yr	2*	Na	Na	Biometric	Geescroft, Rothamsted Experimental Farm, Hertfordshire, England	(Poulton et al., 2003)
Ash-sycamore-hawthorn 120yrs	3.39*	Na	Na	Biometric	Broadbalk, Rothamsted Experimental Farm, Hertfordshire, England	(Poulton et al., 2003)

Measuring above ground carbon stocks in semi-natural native woodlands
Traditionally, allometric models have been used in forestry to estimate above-ground biomass (AGB) and carbon stocks. These size-to-mass models use the relationship between stem diameter and AGB which has been calibrated from destructive measurements of trees which are harvested and weighed alongside measurement of height and stem diameter. However, the suitability of these calibration datasets to accurately assess AGB in native semi-natural broadleaf woodland has been questioned. Recent work by Calders et al., 2022 explored the above-ground biomass carbon stock of Wytham Woods in Oxfordshire, England using terrestrial laser scanning (TLS, sometimes called terrestrial LiDAR). Wytham Woods is dominated by ash, sycamore, hazel and oak and is representative of the structural complexity found in UK semi-natural broadleaf woodland. The TLS method non-destructively generates high resolution 3D models of trees which can be used to calculate tree volume. This study scanned 835 trees and found that the carbon stock of the AGB was 194 t C ha^{-1}. This was 1.77 times (nearly 80%) higher than the carbon stock estimated by traditional allometric models. The authors suggest that this discrepancy is as a result of the calibration dataset for allometric models being biased towards smaller, younger trees that aren't representative of and the structural complexity and plasticity found in semi-natural woodlands. This was highlighted by the authors finding that 50% of the AGB was found in less than 7% of the largest trees (all of which were larger than trees harvested in calibration datasets for allometric models).

This work indicates that we may be significantly underestimating the above-ground carbon stock of native semi-natural woodlands and highlights the significance of protecting these incredibly valuable habitats for nature and climate change mitigation. More work is needed to understand how we can utilise emerging technologies to better understand woodland carbon stocks and update models to better represent the diversity of tree and woodland structure we see in the UK.

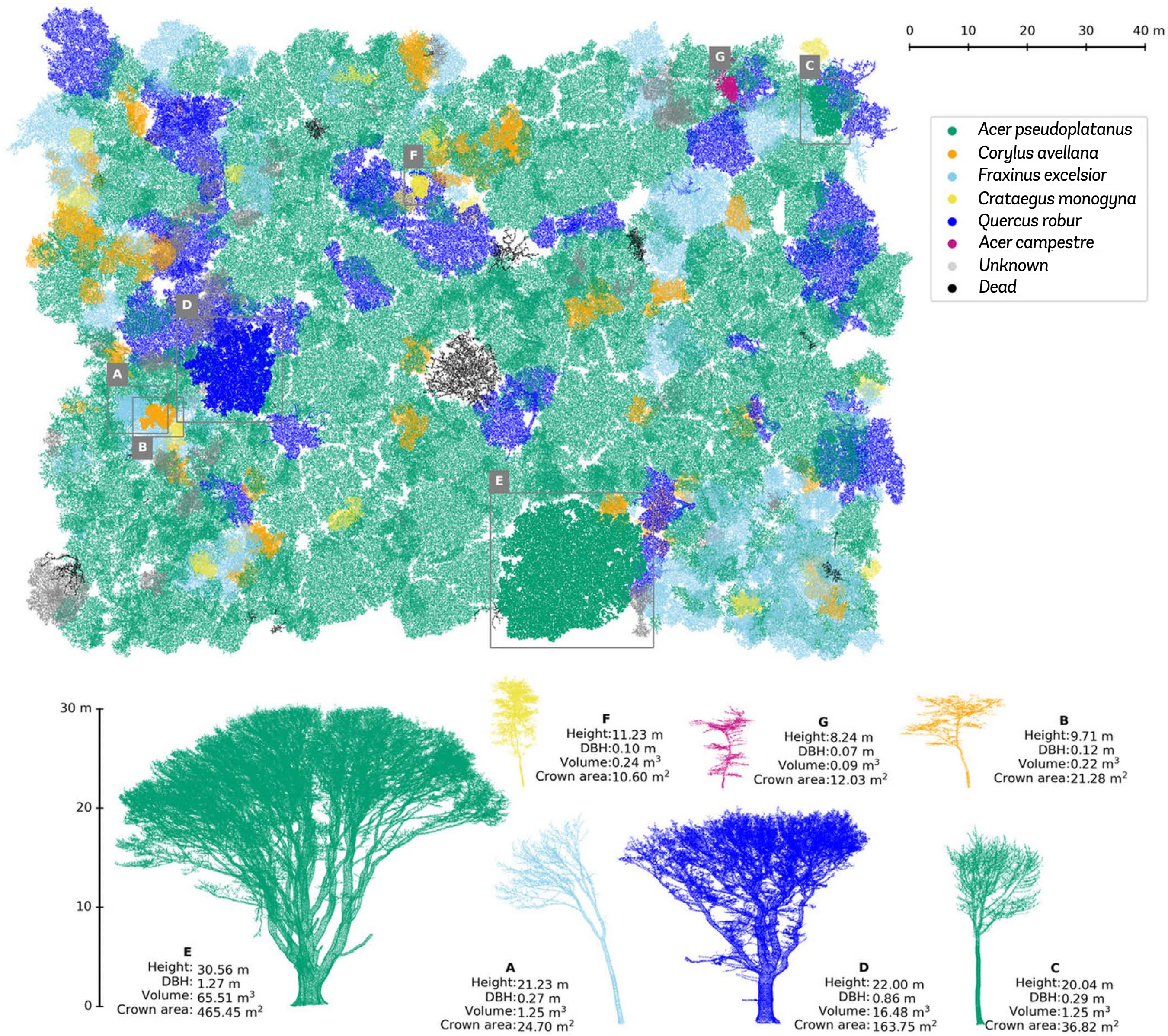

Figure 4. 3D TLS data. (top) top-of-canopy view of 835 trees in the 1.4 ha study area coloured by species. (bottom) side view of individual trees. Reproduced with permission from Calders et al., 2022.

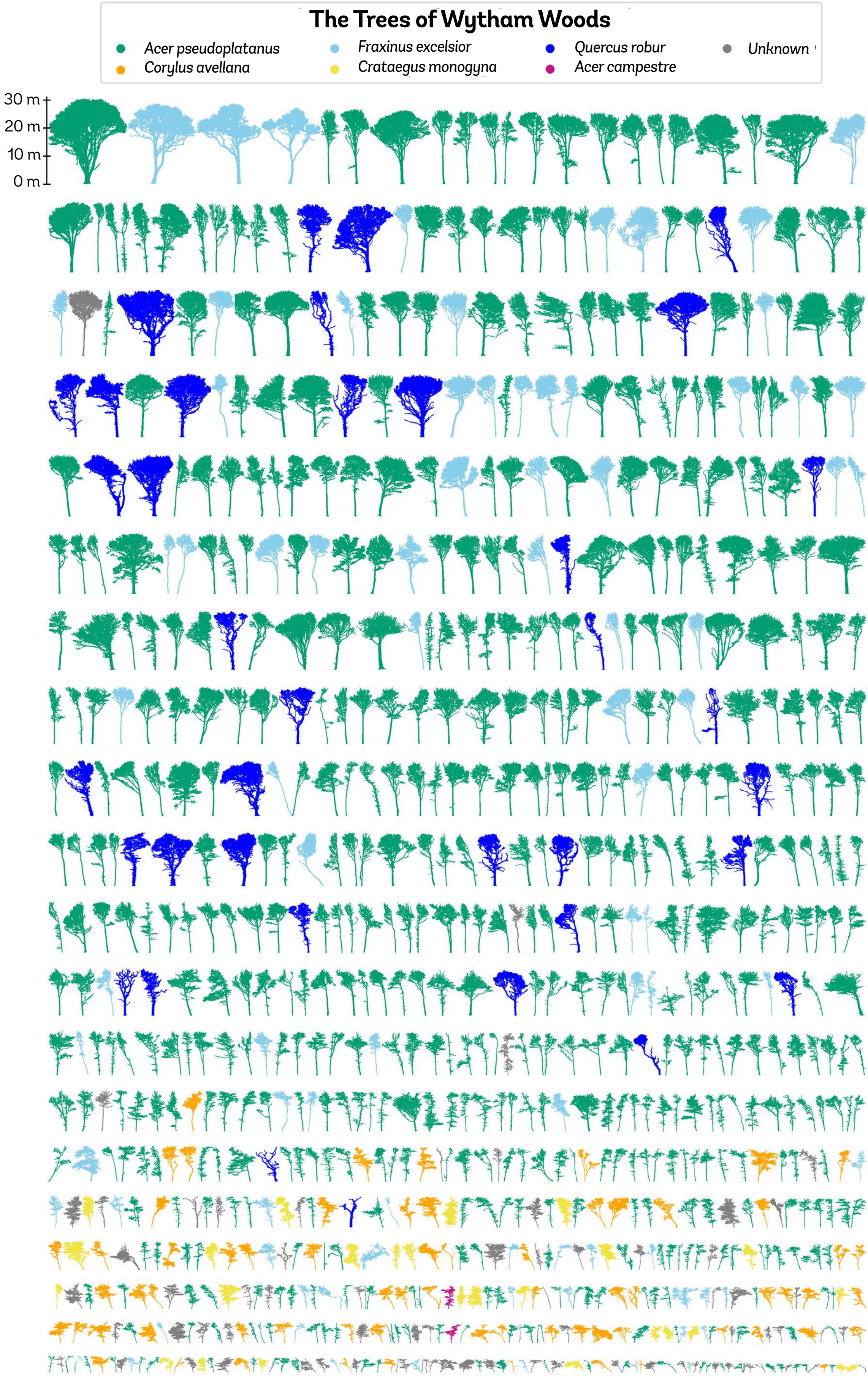

Figure 5. Variation in above-ground structure expressed by 835 trees. Reproduced with permission from Calders et al., 2022

Conclusion

Woods and trees play a significant role in the carbon cycle and the UK's climate change mitigation and net zero strategy. Existing woodland cover stores huge amounts of carbon in both tree biomass and in forest soils. Protecting and strengthening the stability of these carbon stocks needs to be prioritised as declining condition, pests and diseases and the impacts of climate change may lead to significant losses of carbon. New woodland creation has the potential to significantly contribute to atmospheric carbon removals. However, we must balance carbon objectives alongside other priorities such as nature recovery, climate change adaptation and resilience and ecosystem services - focusing solely on carbon outcomes has the potential to lead to negative outcomes for these other objectives.

While the carbon storage and sequestration potential for existing and new woodlands is significant, we also need to recognise that it represents a modest proportion of the political, industrial, and land use changes required to achieve net zero targets and reduce the worst impacts of the climate crisis. Carbon dioxide removals from woods and trees only have the capacity to offset unavoidable residual GHG emissions following effective decarbonisation and emissions reductions measures. Land use and other nature-based measures are no substitute for the immediate, certain and readily quantifiable impacts of a fossil fuel abatement-first approach.

Woodland creation, protection and restoration has the potential to deliver a host of co-benefits for nature recovery, ecosystem services, climate change adaptation, and long-term carbon sequestration and storage, and is a crucial action in respond to a rapidly changing world.

Evidence gaps

- Emerging evidence suggests above ground carbon stocks in structurally complex semi-natural woodlands may be greatly underestimated by traditional accounting methods. We need to better understand these estimates especially for semi-natural native broadleaf woodland which is underrepresented in growth models.
- Carbon stocks in deadwood, woody debris, understorey vegetation and leaf litter are less certain than in tree stems and woodland soil, which should be addressed in future assessments.
- There are only a few woodland stand level studies that directly measure net ecosystem carbon cycling. This limits our ability to understand the impact of environmental and management factors that may alter the carbon balance of these systems.
- The relationship between woodland ecological condition and carbon cycling is poorly understood. In particular, understanding the impact of ecological condition on the stability of carbon pools would greatly inform land management decisions.
- There is lots of uncertainty around the stability of different carbon pools. In particular, the stability of different soil organic matter fractions and the impact of land use and environmental variables is poorly understood.
- The residency time of carbon in different pools is complicated and challenging to examine, however, it is important that we prioritise the

sequestration of carbon into pools that remain out of the atmosphere for long periods of time – certainly as long or longer than the mean residency time of CO_2 in the atmosphere. Our understanding of this is quite limited and estimates can vary significantly.

- Most of the work investigating the role of woods and trees for climate change mitigation and carbon cycling focuses on the exchange of CO_2 with the atmosphere but considerably less is known about the impact of woods and trees on other GHGs such as methane (CH4) and nitrous oxide (N_2O) emissions. Woods and trees also emit volatile organic compounds (VOCs) which can alter atmospheric chemistry and the concentration of GHGs – this is also poorly represented in climate models.
- Alongside the impact of woods and trees on altering GHG dynamics, land use change to woodland cover may also modify land surface reflectance (albedo) which intern could increase the amount of solar radiation that is absorbed, thus increase global warming or decreasing the net cooling effect of carbon sequestration by woodlands. It has been suggested that the impact of this may be more pronounced at northern latitudes. This is currently poorly understood and not included in many climate models of land use change.

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Key definitions

Carbon: a chemical element that forms organic compounds (derived from living organisms and their processes) or inorganic compounds (derived from chemical processes or metabolic processes of living organisms such as respiration producing CO_2).

Carbon dioxide (CO_2): the most important greenhouse gas, released from natural sources such as decomposition and respiration, as well as anthropogenic sources such as the combustion of fossil fuels, industrial processes and land use. Carbon and CO_2 are often used interchangeably when talking about GHG emissions, but it is important to be clear on the difference due to the different molecular masses of each. A carbon atom has a mass of 12 amu (atomic mass units) whereas a CO_2 molecule has a mass of 44 amu; the difference in mass is from the two oxygen atoms in the CO_2 molecule. To convert between CO_2 and carbon, you multiply by the ratio of their masses. Therefore, 1 tonne of carbon = 1 x (44/12) = 3.67 tonnes of CO_2. Conversely, 1 tonne of CO_2 = 1 x (12/44) = 0.27 tonnes of carbon.

Carbon dioxide equivalent (CO2e): a metric used to compare different greenhouse gases based on their global warming potential relative to the amount of carbon dioxide with the same global warming potential.

Global warming potential (GWP): the contribution of a greenhouse gas to absorb energy over a given period of time, usually 100yrs, relative to carbon dioxide.

Carbon sink: an ecosystem is a carbon sink when carbon inputs are greater than emissions.

Carbon source: an ecosystem is a carbon source when carbon inputs are smaller than emissions.

Carbon stock: the amount of carbon contained in a system or its components at a specific time point.

Carbon pool: describes the different components of a system that store carbon.

Carbon/GHG flux: the rate of exchange of a GHG between different pools. Usually between the atmosphere and an ecosystem or component of an ecosystem such as the soil or plants. Usually expressed as a mass of GHG per unit time per area (for example tonnes of CO_2 per hectare per year t CO_2 ha^{-1} yr^{-1}).

Carbon sequestration: usually refers to a process in which carbon is removed from the atmosphere and enters a carbon stock. The term doesn't not necessarily refer to the long-term storage of removed carbon as sequestration can be reversed.

Gross primary productivity (GPP): the amount of carbon dioxide that is assimilated via photosynthesis into organic compounds.

Ecosystem respiration (Reco): the total amount of organic carbon that is released from an ecosystem via autotropic (plant) and heterotrophic respiration (e.g. decomposition by bacteria and fungi).

Net ecosystem productivity (NEP): the difference between the total amount of carbon fixed via photosynthesis (GPP) and the total amount of carbon released via ecosystem respiration; therefore, NEP describes the total accumulation of carbon by an ecosystem. An ecosystem is a carbon sink when the rate of GPP exceeds the rate of ecosystem respiration.

Units

Unit	Tonnes (t)
Kilogram (kg)	0.001
Tonne (t)	1 t
Kilotonne (Kt)	1,000 t
Megatonne (Mt)	1,000,000 t
Gigatonne (Gt) / Petagram (Pg)	1,000,000,000 t

Tree pests and pathogens

Author: Rebecca Gosling

Abstract

Introduced pests and pathogens ('pests') have the capacity to cause widespread impacts on, and losses of our trees, as witnessed with Dutch elm disease and ash dieback. The first edition of ***State of the UK's Woods and Trees*** reported a significant rise in the incidence of serious pest introductions post 1990. Utilising existing datasets of plant pests, we determined that the UK hosts 121 pests of our native tree species which are either introduced or have uncertain origin. It is unknown how many trees the UK loses each year to these pests. Statistics from control of *Phytophthora ramorum* suggest the losses could be high, and management costs are staggering. An estimated £919.9 million is spent each year in the UK on managing only six pests (Eschen et al., 2023). International trade is considered a key driver of these introductions, especially alongside climate change which is predicted to aid pest arrival and/or establishment in the future (Potter and Urquhart, 2017). However, these drivers are complex and do not act in isolation, often creating an unclear picture as to how pests might be introduced into our ecosystems in the future. Further complexities, such as the prediction that many pests are still unknown to science, missing data on the international distribution of many pests, and novel interactions occurring between hosts, pests and vectors, adds significant uncertainty when predicting how and when new pests might arrive. Our trees and woods are precious, providing essential services for us and wildlife, and prevention is better than cure. To protect our woods and trees we must have effective biosecurity systems for imported trees, and we must grow the trees we need here in the UK.

Background

Our ecosystems consist of myriad organisms, each carving out its survival in a unique way. Ecosystems consist not only of large enigmatic species, but also small invertebrates and microorganisms such as fungi and bacteria. These organisms may not always be appreciated, as they eat our favourite plants or cause disease within treasured trees. However, they are all essential to a functioning ecosystem where they act as recyclers and evolutionary forces driving forward natural selection through the 'evolutionary arms race'. Our trees and woodlands play host to a variety of pests and pathogens (or simply 'pests'), most of which are native, meaning they have co-evolved with the other species they live alongside and have their 'roles to play'.

Across the globe, tree pests are moving from their native ecosystems, into new areas where they are considered non-native or introduced. These movements are often attributed to the movement of plants and plant products around the world, which can carry with them these unwelcome guests (Potter and Urquhart, 2017). Introduced pests can quickly become significant threats to our trees, and some have catastrophic consequences. Our trees did not coevolve with these newer arrivals, therefore may lack natural defence mechanisms against them. Additionally, these pests may find themselves with reduced natural enemies, for example, predators which might otherwise aid in population control (Keane and Crawley, 2002). In such scenarios, the introduced pest can quickly proliferate and lead to significant losses of our trees which can be felt across the UK. This has already been observed with two native tree species, ash and elm, which have suffered extensive damage due to two fungal pathogens, ash dieback (*Hymenoscyphus fraxineus*) and Dutch elm disease (*Ophiostoma novo-ulmi*) respectively. Although initially recorded in 1910, the first Dutch elm disease epidemic did not record widespread losses within the UK, however this changed when a more aggressive form of the fungus arrived in the 1960s (Brasier, 1996). Within a decade of the arrival of this new Dutch elm disease fungus, an estimated 20 million elm trees had died (Brasier, 1996). Later estimates in the 1980s suggest that around 30 million elms, nearly the entire UK elm population, were lost (Brasier, 1996; Potter et al., 2011). The ash dieback epidemic is still playing out; the latest Bunce survey reported 49% of ash woodlands displaying signs of the disease (Smart et al., 2024). Mortality within ash woodlands has been recorded as high as 85% of all ash trees, with maximum mortality typically reached within 10-15 years after exposure to ash dieback (Coker et al., 2019). This rapid and widescale loss of trees can significantly impact the valuable ecosystem services that trees and woodlands provide, some of which may never recover (Boyd et al., 2013). For example, the replacement of ash with another tree species might restore carbon storage potential, but the unique biodiversity support that ash provides as a species is irreplaceable (Boyd et al., 2013; Mitchell et al., 2017). Ecosystem services such as the removal or interception of air pollutants can also be reduced when large scale tree losses occur. In North America, bark-boring pest outbreaks including outbreaks of the introduced emerald ash borer (*Agrilus planipennis*), have been linked to increased respiratory and cardiovascular illnesses (Donovan et al., 2013; Jones, 2021). Introduced pests have wide-reaching consequences.

Climate change may shift this dynamic further, creating additional stress within trees that allows previously unproblematic pests to become increasingly damaging (Hartmann et al., 2022; Pautasso et al., 2012). Projected climate change trends include warmer, wetter winters and hotter, drier summers, as well as increases in the frequency and intensity of extreme weather (Met Office, 2022). It is thought that this could alter the severity of pests through direct effects on the pests' developmental and survival prospects, indirectly by impacts on their natural enemies, and via changes in the host tree such as increased stress (Krokene, 2015; Wainhouse and Inward, 2016). Because of their generally short life cycles, great reproductive potential, sensitivity to changes in temperature and, in many cases, great capacity for dispersal, even moderate changes in climate could have significant rapid impacts on the distribution and abundance of many tree pests (Wainhouse and Inward, 2016).

This chapter will explore the current understanding and data behind the incidence of introduced tree pests in the UK. To protect our trees and woods from these threats it is important to understand their presence and distribution, the drivers behind their introductions, their current impact and future risks. The questions the chapter will seek to answer are:

1. How many introduced pests are present in the UK? How many of these pose

a serious risk? And how many of these use our native tree species as a host?
2. What do we know about the distribution of tree pests in the UK? Have any introduced pests spread across the whole country?
3. What are the key pathways of pest introductions?
4. How many trees do we lose from introduced pests each year? And what is the economic impact of this?
5. What do we know about the future risk posed by further pest introductions?

Methods

This chapter utilises data from multiple sources to present the presence, distribution, drivers and impact of introduced tree pests. Figures were created in R, using the package ggplot (R Core Team, 2023; Wickham, 2016).

Question one: How many introduced pests are present in the UK? How many of these pose a serious risk? And how many of these use our native tree species as a host?

The number of serious tree pests in the UK was included in the first *State of the UK's Woods and Trees*, Figure 3.4.1, 'timeline of when pests and diseases were first reported as causing serious issues with particular host species in the UK (1950–2018)' (Reid et al., 2021). This graph was updated and adapted based upon new introductions and a revised methodology to allow replication in the future. The data was sourced from Forest Research and Defra (Defra, 2024a; Forest Research, 2024). Key introduced pests of trees were identified using the following criteria:

- Widespread distribution. Defined as not present in only one county or only found indoors.
- Common tree species are the main hosts, including all native species, naturalised species such as sycamore (*Acer pseudoplatanus*) and sweet chestnut (*Castanea sativa*), and commonly planted forestry species such as Sitka spruce (*Picea sitchensis*).
- The impact of the pest has been recorded and is considered damaging across its UK range.
- Information is available to assign the year of the first record.

A database including all introduced tree pests in the UK is not known to be available. Therefore, an analysis was conducted to create a new database containing introduced pests which use native tree species as a host. However, for some pests, especially viruses and phytoplasma, information on their introduced status (i.e. their origin) is not available as it would be extremely difficult to determine. Therefore, the database created includes pests where introduced status is uncertain so these pests have been collectively labelled 'detected pests' rather than 'introduced pests' (supplement one). Pests of interest were identified using the following methods:

- Data from the UK Plant Health Risk Register was exported on 2 May 2024. The data was filtered to pests present in the UK, and those hosted by native tree species, or a genus which includes a native species, for example '*Quercus spp.*' (Defra, 2024a).

- The CABI Horizon Scanning Tool (free version) was used, selecting United States/New Hampshire as the area at risk from UK/NI (CABI, 2024a). Specifying habitats 'natural forests' and filtering to only show 'plant pests' and 'invasive species'. United States/New Hampshire was selected as a region with a similar climate to the UK, hosting temperate woodland, but with limited duplication in tree pest presence to allow the tool to identify pests recorded as present in the UK. The data was then manually filtered to identify introduced pests of trees.

The pests for inclusion were selected if they had confirmed presence in the UK. The following information was collected for each pest.

- Common name
- Latin name
- Type of pest, for example, bacterium, invertebrate, virus
- Host species of interest
- Presence in UK
- Statutory or regulated status in the UK and the EU.

From the resulting data, the number of known and reported detected pests of trees present in the UK could be identified, alongside the number which have regulatory interest.

Question two: What do we know about the distribution of tree pests in the UK? Have any introduced pests spread across the whole country?

The distribution of many tree pests is unknown, therefore presenting distribution was restricted to pests where a reliable dataset was available. Data was available for three pests, ash dieback, oak processionary moth (*Thaumetopoea processionea*) and *Phytophthora ramorum*. Ash dieback data was downloaded from the Forestry Commission open data portal on 2 February 2024, under an open government licence v3.0 (Forestry Commission, 2024a). Oak processionary moth data was provided via email by the Forestry Commission Plant Health Team on 16 February 2024. *Phytophthora ramorum* data was provided on a country level by England, Scotland, and Wales, but was unavailable for Northern Ireland. Data for England was downloaded from the Forestry Commission open data portal on 31 January 2024, under an open government licence v3.0 (Forestry Commission, 2023a). Data for Wales was provided by email from Natural Resources Wales on 7 December 2023, under an open government licence v3.0. Data for Scotland was provided by email from Scottish Forestry on 19 June 2024.

Question three: What are the key pathways of pest introductions?

The drivers of pest introductions are presented for two widely reported influences, international trade and climate change. A scoping literature review was conducted to understand the evidence behind each potential driver, to identify initial gaps in understanding and to identify any additional drivers. The search was conducted using Google Scholar to find both published peer-reviewed literature and grey literature. The following search terms were used:

- 'forest*' OR 'wood*' OR 'tree*' AND 'pest' OR 'disease' AND 'international trade' OR 'import*'
- 'forest*' OR 'wood*' OR 'tree*' AND 'pest' OR 'disease' OR 'pathogen' AND

'climate' OR 'temperature' OR 'weather'

Reference lists were checked to see if they contained any additional relevant studies. Studies from outside the UK were included if they fall into similar climatic and ecological conditions when considering the impacts on climate change.

International trade data for imports of trees is published by Defra annually. Data on weight, number, and value of trees, shrubs and bushes imported into the UK has been published for 2016 – 2022 (Defra, 2021, 2023). Data was also used from the Animal and Plant Health Agency (APHA) detailing interceptions made on plant imports weekly from March 2021 – June 2024 (Defra, 2024b). The data was filtered for the intercepted pests that include common tree species as hosts (native, naturalised, and commonly planted forestry species).

Question four: How many trees do we lose from introduced pests each year? And what is the economic impact of this?

Data on tree losses due to introduced pests in the UK is not recorded. However, data is available and published for the issued Statutory Plant Health Notices (SPHNs) which aim to control pests of regulatory importance. Data on the hectares covered by Statutory Plant Health Notices was extracted from Forestry Statistics 2023, and presented alongside planting data for context (Maxwell, 2023). To ensure a proportionate representation the data was converted into a total percentage of land area for the country rather than a raw value (England, Scotland, Wales, Northern Ireland).

The cost of managing introduced invasive species in the UK was published in 2023 (Eschen et al., 2023). Using data from Table 5 in Eschen et al., the estimated cost of managing the six tree pests included in the study could be calculated.

Question five: What do we know about the future risk posed by further pest introductions?

To determine the future risk from further tree pest introductions, data was extracted from the UK Plant Health Risk Register on 2 May 2024 (Defra, 2024a). The data was filtered to pests 'absent' from the UK, then further filtered to those who list a native species as a 'major host', or a genus which includes a native species, for example 'Quercus spp.'. Resulting data was filtered according to the 'relative risk rating (mitigated)'. Low risk was assigned as 0-29, medium is 30-59 and high risk is 60 and above. This follows a similar methodology to the risk register that includes 'red' species as 60 and above, orange, and yellow as 30-59 and blue and green as 1-29. Any data without a risk rating was discounted.

Results

Question one: How many introduced pests are present in the UK? How many of these pose a serious risk? And how many of these use our native tree species as a host?

The first edition of *State of the UK's Woods and Trees* reported a significant rise in the incidence of serious pest introductions post-1990. Since this report, two more serious pests of trees have been found in the UK. This includes *Phytophthora pluvialis* which was first found in Cornwall in 2021, and plane lace bug first found in London in 2024 (Forestry Commission, 2023b). This *Phytophthora* has since been recorded across the west of Britain, indicating that it was likely introduced multiple years before its detection (Forestry Commission, 2023b). Due to the revised methodology, ***Phytophthora siskiyouensis, Phytophthora pseudosyringae, Xylosandrus germanus,*** and box tree moth (***Cydalima perspectalis***) were also added to the dataset (Figure 1).

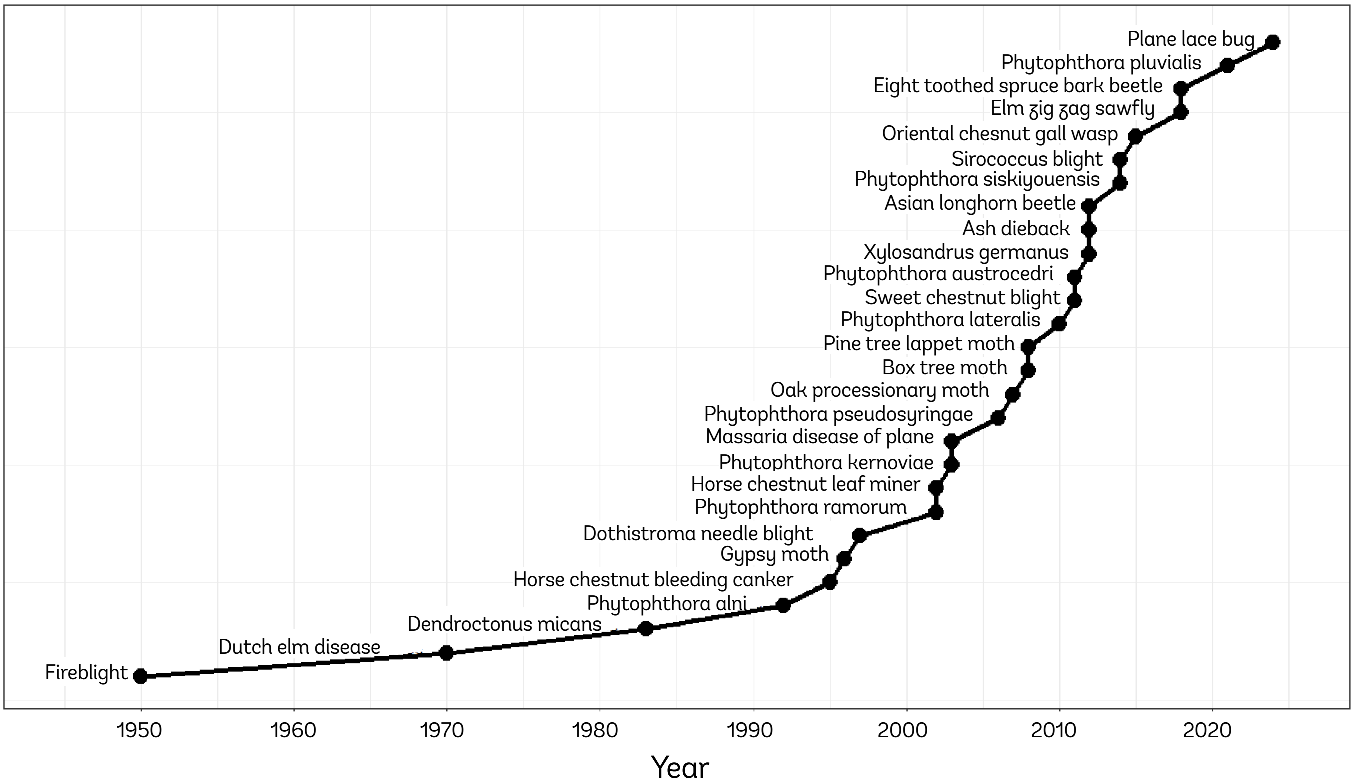

Figure 1. Serious pests of trees introduced into the UK by year, since 1950 (Defra, 2024a; Forest Research, 2024).

Furthermore, many other pests are present which may not be considered serious but have still been identified and reported. Results from new analysis, which created a database of detected pests in the UK which can use native tree species as hosts, demonstrates that the UK hosts 121 detected pests of native trees (supplement one). Many of these have been introduced, however, for some pests (such as viruses) assigning their native range was not possible. This includes 18 of the 28 pests in Figure 1. Additionally, this analysis revealed that every native tree species in the UK has the potential to host one of these pests. A total of 59 of the threats are of European importance, meaning they are listed in legislation as either quarantine or regulated non-quarantine pests (Regulation (EU) 2016/2031).

Question two: What do we know about the distribution of tree pests in the UK? Have any introduced pests spread across the whole country?

Data was available on the distribution of ash dieback, *Phytophthora ramorum* and oak processionary moth (*Thaumetopoea processionea*). This is presented in 10km squares (Figure 2). Whereas none of these pests have been recorded across the full extent of the UK, this is in part due to distribution of the hosts.

Ash dieback is widespread across the UK, the gaps within this dataset, especially in northern Scotland, are reflective of reduced ash presence within this region. *Phytophthora ramorum* is more prevalent on the west coast due to the more amenable climate for this pathogen, and oak processionary moth is slowly spreading out from its original outbreak in London. Between the three introduced pests there is almost full coverage of the UK.

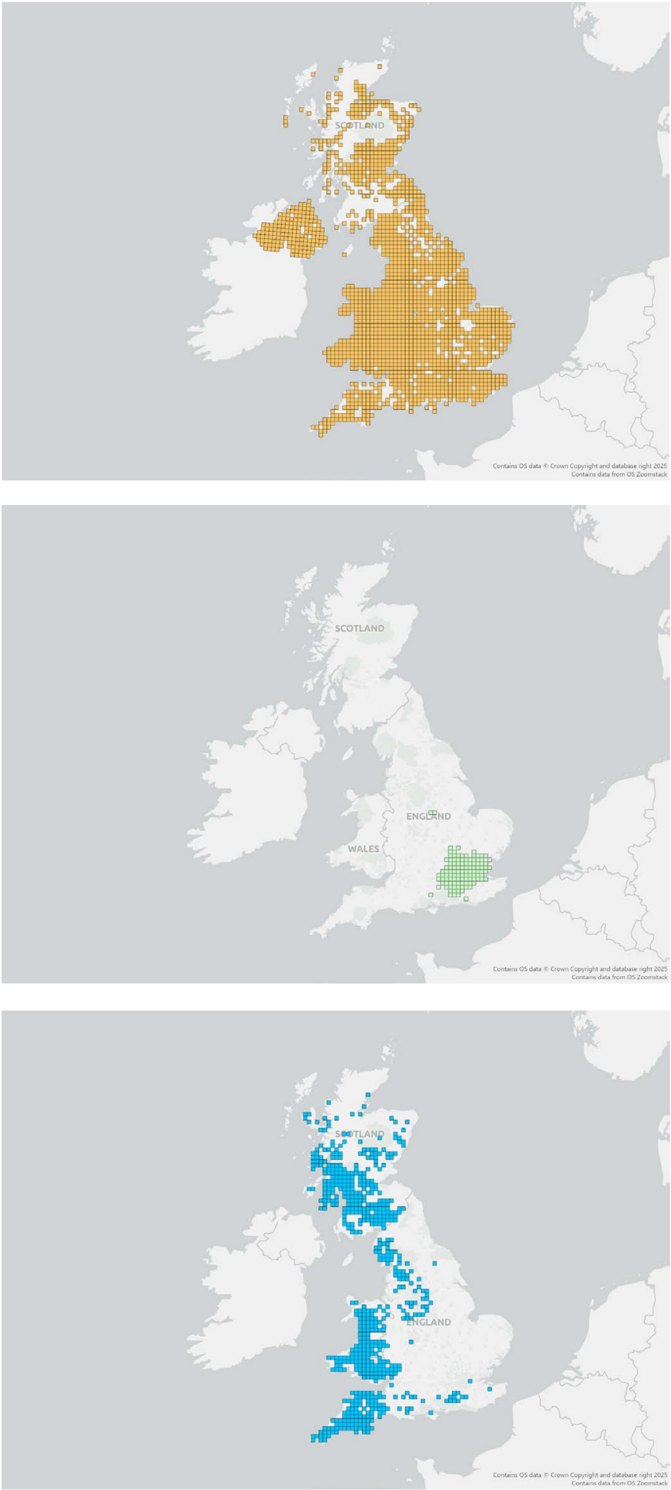

Figure 2. 10km grid squares displaying the UK distribution of *Phytophthora ramorum* (blue), ash dieback (orange) and oak processionary moth (green). Credits: Contains Natural Resources Wales information © Natural Resources Wales and database right. All rights reserved. Forestry Commission Plant Health Team, Defra, GeoData, University of Southampton. Forestry Commission. Scottish Forestry.

Question three: What are the pathways of pest introductions?

International trade and climate change were the most consistently cited drivers of international tree pest introductions (Boyd et al., 2013; Eyre et al., 2013; Ghelardini et al., 2016; Liebhold et al., 2012; Potter and Urquhart, 2017; Roques, 2010; Santini et al., 2013, Sikes et al., 2018; Woodward et al., 2022). Other drivers cited include tourism and travel, new interactions between pathogens and insects, hybridisation of pathogen species and large monoculture plantations (Boyd et al., 2013; Ghelardini et al., 2016; Potter and Urquhart, 2017).

It is also worth noting that increasing opportunities for introductions and movement of organisms around the world also presents the prospect of novel interactions which can accelerate the ability of pests to cause damage (Burgess et al., 2015; Ghelardini et al., 2016; Santini et al., 2013; Wingfield et al., 2016). This can include new interactions between insects and pathogens, the hybridisation of previously geographically separated pathogens, or new interactions between host trees and pests. A key case study to demonstrate this is Dutch elm disease. The fungus that causes this disease hybridised with a closely related species when it spread across the northern hemisphere, and it is suggested that the gene transfer involved enabled it to become a highly damaging pathogen (Boyd et al., 2013; Paoletti et al., 2005). Dutch elm disease also presents a novel interaction between the pathogen and European elm bark beetles (Scolytus spp.), which became the mechanism for its spread across Europe (Brasier, 1996; Wingfield et al., 2016). More recently in the UK, a novel interaction between host and pathogen has been reported for *Phytophthora pluvialis*. First detected in 2021, it has been recorded on two novel hosts, western hemlock (*Tsuga heterophylla*) and Japanese larch (*larix kaempheri*) (Perez-Sierra et al., 2022, 2024).

International trade

The rise in international trade, particularly since the 1990s with the opening of more trade routes from newly industrialising countries, is attributed to facilitating the increased movement and introduction of tree pests (Boyd et al., 2013; Eyre et al., 2013; Panzavolta et al., 2021; Potter and Urquhart, 2017; Roques, 2010; Woodward et al., 2022). The UK Plant Health Risk Register includes 1,428 plant pests, of which 633 list live plants as a potential pathway to introduction to the UK (Defra, 2024a). Although this includes non-tree hosts, it does demonstrate the prevalence of this pathway. It is estimated that between 1970 and 2004, plant trade was responsible for 89% of invertebrate plant pest introductions into Great Britain (Smith et al., 2007). In the United States, 70% of damaging forest insects which established between 1860 and 2006 likely entered the country on imported live plants (Liebhold et al., 2012). Likewise in Europe, between 1800 and 2008, 57% of introduced pests (in this case exclusively pathogens) were moved through live plants, and 10% by wood products (Santini et al., 2013). Although these studies provide key evidence for this driver of pest introductions, they do have a key limitation. Information on the exact pathway of introduction is unavailable for many pests, so they also use predictions based upon knowledge of the pest's distribution and biology (Liebhold et al., 2012; Santini et al., 2013; Smith et al., 2007). Two projects in Europe reported statistically significant correlations between the number of introduced pests of woody plants and volume of manufactured and agricultural imports for European countries (Roques, 2010). However, the study does recognise the complexity of the situation, and that other factors other than volume are likely influential, including speed of trade, trade routes, products traded and the exporting and importing countries' biosecurity procedures (Roques, 2010). Reflecting that whereas international trade is contributing towards movement of pests around the world, factors influencing this and evidencing the pathway of movement can be complex.

Timber, wood products and wood packaging are also commonly quoted as moving damaging tree pests around the world (Liebhold et al., 2012; Potter and Urquhart, 2017; Smith et al., 2007). Wood packaging, especially when untreated, poses a significant risk for the movement of wood boring pests, such as Asian longhorn beetle (*Anoplophora glabripennis*) (Haack et al., 2010). An outbreak of this pest in Kent in 2012 was suspected to originate from wood packing from a nearby stone importer (Eyre and Barbrook, 2021). The successful eradication of this pest in Kent required the removal of 2,133 trees through a programme costing an estimated £1.9 million (Eyre and Barbrook, 2021). Between 1980 and 2008, 97% of worldwide interceptions of Asian longhorn beetle (*A. glabripennis*) related to movement of wood packaging material from its native range in China (Eyre et al., 2013).

Import data for trees, bushes and shrubs shows that in 2022, 126 million individual trees, bushes and shrubs were imported into the UK, at a value of £259 million (Figure 3). The methodology for this reporting changed in 2022 due to the UK's exit from the European Union, meaning that more imports are now being recorded (ONS, 2022). Therefore, we would expect to see an increase in all these statistics from 2022. However, the number of plants imported decreased in 2022, to 126 million, when compared to 211 million in 2019 and 216 million in 2020. But there was an increase from 2021 which reported 67 million individual plant imports. Whereas the net mass and value increased considerably. The net mass and value of imports was reported as 124 thousand tonnes and £259 million in 2022, representing a 162% and 391% change respectively from 2016 to 2022 (Defra, 2021, 2023). This shift in imports, to less individual plants but with a higher value and net mass, could potentially represent a move towards importing larger more valuable plants, but more research would be required to confirm this.

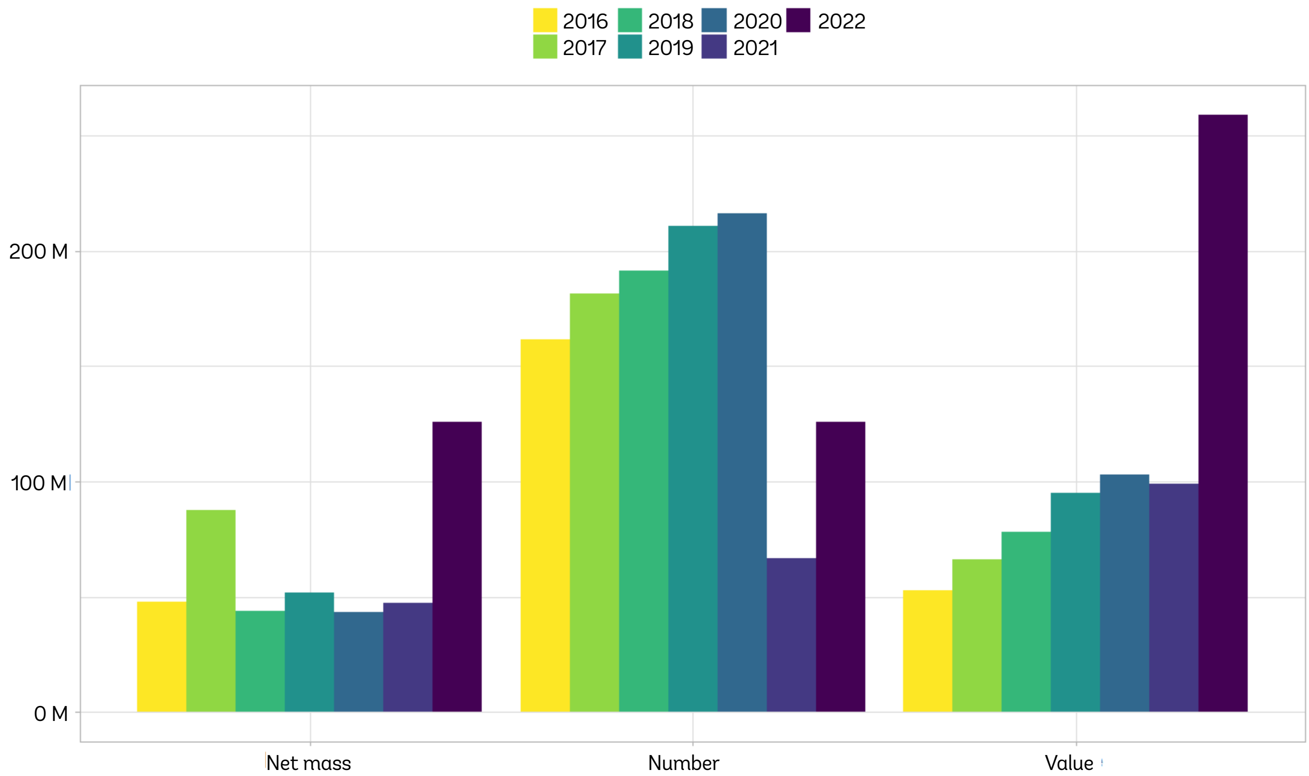

Figure 3. The net mass (kg), value (£) and number of trees, shrubs and bushes imported into the UK annually from 2016 – 2022 (Defra, 2021, 2023).

Imports of trees are considered 'high risk' and are therefore subject to risk-based import inspections. Since 2022, 128 interceptions of tree pests have been reported (Table 1). Significant interceptions include pine processionary moth (*Thaumetopoea pityocampa*), a defoliator of pine with human health risks from its urticating hairs, and Anoplophora species, a genus which includes many highly damaging tree pests (Defra, 2024b). Additionally in 2023, imports of *Rhododendron ponticum*, an invasive plant species which is already causing damage across our woodlands, were intercepted infected with *Phytophthora ramorum* (Defra, 2024b). The increases in interceptions observed may be due to the increased inspection regime that has been phased in since the UK's departure from the European Union.

Table 1. Interceptions of tree pests on imported plants since 2022 (Defra, 2024b). *since 20 March. **until 13 June.

Year	2021*	2022	2023	2024**
Number of Interceptions	8	38	40	42 to June

Climate change as a driver of tree pests and pathogen introductions

Climate is part of a complex web of factors that can influence pest abundance and distribution, including host stress, host availability, phenology, natural enemies, competitors and human mediated movement (Simler-Williamson et al., 2019; Wainhouse and Inward, 2016). This makes it very difficult to show that climate change has specifically and solely aided the arrival and/or establishment of any tree pests. It is not possible to predict the future impact of pests with any precision, but some generalisations may be made based on ecological characteristics of different species. Therefore, this section will use literature that discusses climate impact on all pests and pathogens, native and introduced.

Pathogen ecology is strongly dependent on environmental factors, such as temperature and moisture (Santini et al., 2013). In the UK, climate change is predicted to cause milder and wetter winters alongside increased spring rainfall, which is likely to enhance the survival and infection potential of many tree pathogens (La Porta et al., 2008; Pope et al., 2022; Watts et al., 2015). This is predicted to increase the growth or spore release of many of the common widespread pathogens, leading to increased infection, and may also extend the active period of pathogens, allowing them to cause successive damage to hosts (Harvell et al., 2002; Marcais and Despreȝ-Loustau, 2014). For example, *Phytophthora cinnamomi* is highly dependent on soil water content and mild temperatures, warmer, wetter winters could enhance the ability of this pathogen to cause decline within its hosts, which include many tree species such as beech and oak (Keca et al., 2016; Simler-Williamson et al., 2019). Higher temperatures can also increase sporulation for some pathogens and shorten their lifecycle allowing multiple generations within one season (La Porta et al., 2008). Infection rates of *Heterobasidion* spp. that cause root and butt rot in coniferous trees are predicted to increase under warmer climates due to increased spore production (La Porta et al., 2008; Muller et al., 2014). Changes in climate are predicted to shift and expand the distribution range of some pathogens. Pathogens that have been of importance in southern Europe may spread northward; this migration of pathogens may increase disease incidence when they encounter new hosts or vectors (La Porta et al., 2008).

Climate events causing increased stress in trees are also likely to increase host susceptibility to pests; these include hotter, drier weather causing drought, or flooding causing water logging (Domisch et al., 2020; La Porta et al., 2008; Wainhouse and Inward, 2016). Increased susceptibility to attack is thought to be due to reduced tree defence capacity resulting from lower carbohydrate production during times of stress, reducing the availability of resources for the synthesis of secondary defence metabolites that aid in resisting pests and pathogens (Krokene 2015). For example, drought stress within hosts can increase canker development and mortality caused by the fungal pathogen that causes chestnut blight (*Cryphonectria parasitica*) (Gao and Shain, 1995; Waldboth et al., 2009).

Warmer temperatures are likely to have complex effects on insects, influencing their development rate and life cycles whilst also affecting their natural enemies (Wainhouse and Inward, 2016). The damage caused by aphids and other similar insects are likely to grow with warming temperatures as their reproductive rates are predicted to rise, and drought stress of trees may increase their susceptibility to aphid attack (Wainhouse and Inward 2016). The impact of bark beetles and related insects is also expected to increase due to factors such as heightened frequency of windthrows, drought stress, and for some species, a shorter generation time allowing rapid population growth (Hlasny et al., 2019; Hlasny and Turcani, 2008). While trees are usually well defended against beetle attacks, during endemic phases the sheer number of attacks can also cause healthy trees to succumb

(Biedermann et al., 2019; Kolb et al., 2019; Krokene 2015; Netherer et al., 2021, 2024). Effects upon defoliators are more difficult to predict but likely include an increase in abundance and impact as the number of generations and geographical distribution increases under warmer climates (Wermelinger & Seifert 1998). Ranges of introduced pests may also increase under climate change. For example, in Europe, 186 insect pests of trees and shrubs have been introduced from subtropical or tropical climates, indicating that they can currently survive in warmer European regions such as the Mediterranean, however under a warmer climate it is likely that this range will increase (Roques, 2010). This is predicted for oak processionary moth with Scotland and northern England likely to become more suitable for this pest in future climate scenarios (Godefroid et al., *2020)*.

However, the complex dynamic of a multitude of factors can influence the distribution and abundance of insects, including food resources, natural enemies, competitors and climate (Wainhouse and Inward, 2016). For example, changes in seasonal temperatures are also likely to alter the phenology of trees, potentially resulting in a mismatch in phenology between insects and their host trees (van Dis et al., 2023). For defoliating insects this could mean less food and high mortality of young larvae (van Dis et al., 2023). Similarly for pathogens, powdery mildew (Erysiphe spp.) currently displays phenological synchrony with oak bud burst; a change in phenology could reduce the impact of this pathogen (Marcais and Desprez-Loustau, 2014). It is also recognised that predictions based on climate or regional weather alone may not provide an accurate projection as pests and pathogens also respond to microclimates within the habitats where they live. To improve the estimates of these risks, the Met Office, the University of Exeter, Forest Research and Kew Gardens are currently monitoring microclimates at five sites in southern England and results are expected to provide more insight into microclimate influences (Met Office, 2024).

In summary, although climate impacts on pest and pathogen interactions is complex, using predictions based upon both native and introduced pest ecology, it is likely that climate change will aid the arrival and/or establishment of new tree pests in the future. However, more research is required to provide more detailed information on which pests and pathogens might be most influenced by climate change, alongside which hosts might also be most impacted causing increased susceptibility, to allow risk-based mitigation measures.

Question four: Do we know how many trees we lose from introduced pests each year? And what is the economic impact of this?

Case study: *Phytophthora ramorum* at Wentwood

In 2002, a previously unknown plant pathogen was discovered on a batch of imported Viburnum plants in England and was named *Phytophthora ramorum.* This highly pathogenic algae-like pathogen, likely native to Asia, has a large host range of more than 150 plant species and has gone on to have significant impacts across Great Britain. In 2009 it was found that *P. ramorum* had jumped host to larch, which, it transpires, is very susceptible. In efforts to control this pathogen and slow its spread, the Forestry Commission and Scottish Forestry conduct regular surveillance using helicopters to fly over large tracts of forests to identify the disease symptoms. Should symptoms be found, and a follow up survey on foot confirm presence of *P. ramorum*, the landowner will be served a statutory plant health notice, frequently referred to as an SPHN (unless the trees are in South West Scotland which is a designated management zone, which means it has been acknowledged that the disease is too far advanced to control the spread in this region). This requires either felling, or in some cases, killing the trees through ring barking or stem injections. However, the preferred and most common method to ensure the trees are removed before the pathogen can sporulate and spread further, is felling. The SPHN would typically require all larch species within a 100-metre buffer from the infection to be felled. Therefore, findings of *P. ramorum* in larch on plantation on ancient woodland site (PAWS) sites can be devastating for restoration efforts, requiring large scale felling of trees that would otherwise be gradually removed to allow the ecosystem to recover slowly.

ROB DAVIES

ROB DAVIES

Ramorum @ Wentwood

Wentwood is a Woodland Trust-owned woodland just outside Newport in South Wales. The 353-hectare site is part of the largest remaining block of ancient woodland in Wales, a remnant of the continuous forest that would once have stretched from the River Usk to the River Wye. It has a rich history. Wentwood is mentioned in ancient texts, such as the 12th century Book of Llandaff, and contains important ancient monuments including bronze age burial mounds. The two world wars resulted in large-scale felling of the broadleaf trees which were replaced with non-native conifer species to produce timber quickly. In Wentwood, this resulted in large plantings of Douglas fir, Norway spruce, and larch.

The Woodland Trust purchased this PAWS site in 2005, with a view to gradually restoring the site to an ancient semi-natural woodland through slow conifer removal. This process would typically take around 30 to 40 years, to slowly thin the blocks of conifers, allowing light onto the woodland floor and regeneration of ancient woodland flora. The surviving ancient woodland species are often adapted to the conditions of a woodland, preferring seasonal shade in sheltered conditions. Removing large numbers of trees in one go would allow bramble and bracken to take over. Dramatic management interventions such as this can also damage ancient woodland features, exposing areas of deadwood to desiccation and windblow of the remaining trees due to their loss of shelter. Therefore, restoring an ancient woodland takes time and patience and is a gradual process to allow the woodland to return in its own time.

Unfortunately, due to *P. ramorum* and the associated statutory plant health notices (SPHNs), demonstrating PAWS restoration best practice by gradually thinning larch from this woodland was no longer an option. Over the past 10 years Wentwood has been issued with a series of SPHNs for *P. ramorum* on larch. These SPHNs have a legal basis, requiring a landowner to carry out certain management on their land. At Wentwood this resulted in clear felling of over 146 hectares of larch-dominated woodland, which is 41% of the woodland. One SPHN alone required the felling of 60 hectares, often on very short timeframes to prevent sporulation and spread of the pathogen in the autumn. This disruptive large-scale change to the woodland presented challenges in ensuring its continued restoration into an ancient semi-natural woodland.


ROB DAVIES


ROB DAVIES

Direct seeding and fenced.

The Woodland Trust was also conscious that any new plantings or regeneration needed to be protected from deer browsing, another key threat to woodland restoration, especially when planting large areas (see deer chapter). Multiple methods have been explored and utilised on this site, including trees in plastic-free tubes, planting at higher densities without tree tubes, direct seeding with deer fencing and encouraging natural regeneration with enrichment planting. Whereas this may seem like a good way to 'fast track' restoration, and the Woodland Trust has worked hard to recover this woodland and demonstrate a variety of techniques in doing so, this is not the ideal situation for PAWS restoration. To retain optimum ecological integrity and prevent risks to important remnant features, restoration should be conducted on much larger timescales in a much more delicate and bespoke manner. The introduction of *P. ramorum* has prevented this for many PAWS woodlands, like Wentwood, across the UK. As demonstrated earlier in this chapter, the pathogen continues to spread, and recently SPHNs have also been received for sweet chestnut infected with *P. ramorum* in South West England. This demonstrates how, not only does the pathogen itself impact our woodlands, but so does the resulting control action as the authorities attempt to slow the spread.

Case study: Urquhart Bay

Urquhart Bay Wood is a Woodland Trust site nestled in a sheltered cove on the north side of Loch Ness adjacent to the popular tourist hotspot of Drumnadrochit, where people are drawn by the hope of spying the very shy resident of the nearby loch. It is a Site of Special Scientific Interest (SSSI) and a Special Area of Conservation (SAC), designated for its importance as a floodplain woodland, one of the largest and most intact examples in the UK. It is, however, under attack from all angles.

The woodland has changed radically in the last 15 years. In the mid 2000s, the canopy would have been dominated by ash with wych elm, sycamore and alder throughout. To reduce the threat from non-native sycamore shading native trees and ancient woodland ground flora, these were removed, creating space for ash and alder to quickly infill, along with a healthy shrub layer of bird cherry and hazel. Dutch elm disease swept through, further reducing the canopy's diversity to ash and alder, dramatically increasing the standing deadwood component of the woodland structure.

In 2019, an Observatree volunteer unfortunately confirmed the inevitable arrival of ash dieback in a few of the younger trees. With the low lying nature and high humidity of the woodland, the disease spread ferociously. It was quickly apparent that there would be a huge amount of work in safely reducing tree risk while retaining some form of canopy for as long as possible and not compromising the important habitats for lichens and mosses that rely so heavily on mature ash. The Trust began reducing the volume of standing high risk elm and reducing the canopies of some ash annually to try to retain as much as possible, for as long as safely possible, while leaving ash inside the wider wood to naturally decline and fall.

2019

2020

2021

2022

Progression of ash dieback recorded at Urquhart Bay over four years by Observatree volunteer David Slawson

With the ash canopy declining, the increased light has allowed a cohort of young sycamore to flourish, which has received a reprieve as sycamore can be a suitable surrogate for many of the lower plant species currently reliant on ash, resulting in the future canopy of sycamore and alder. Or does it? A new unwelcome arrival 2024 is suspected *Phytophthora alni*, which can be fatal for alder and will threaten this ravaged woodland even further, encouraging us to look at planting other suitable natives to increase the woodland diversity for the future.

An opening canopy causes growing concern for the ground flora, largely dominated by bluebell, at this site. A whole assemblage of invasive plants are taking advantage of these conditions and are doing so right up the catchments of the rivers that eventually find themselves in Urquhart Bay. Only through collaboration with our neighbours, agencies, and community will this meaningfully be tackled. The one saving grace for this woodland is that browsing by deer is low, allowing young trees to establish. Even in a site so heavily impacted by disease and invasive plants, there is a ray of hope if you look hard enough.

About Observatree

Observatree is an important part of the UK's response to tree pests and pathogens. The project, led by Forest Research, includes multiple governmental and NGO partners, including the Woodland Trust. Observatree recruits and trains a network of 200 volunteers who aid biosecurity efforts by acting as an early warning system for serious tree threats. These volunteers are on the lookout for 24 priority pests and pathogens, some which are present in the UK, where they aid in monitoring spread and distribution, and some not. For those pests and pathogens not yet present, Observatree volunteers are essential eyes on the ground, aiming to detect any new introductions as soon as possible. By submitting high quality accurate reports straight to governmental scientists via the TreeAlert portal, the flow of information is rapid and lands right where it needs to. Observatree is highly regarded, both within the UK and internationally, as a model for public engagement and early warning. Find out more at observatree.org.uk.

ROSS WATSON

Ash dieback at Urqhart Bay

Tree losses

Statutory plant health notices (SPHNs) are a legal instruction to the landowner, issued to both eradicate and control tree pests. SPHNs are currently issued to control the spread of P. ramorum which usually requires the felling of infected and surrounding larch (Larix spp.). The data shows Wales has the greatest percentage of land area with SPHNs applied over the past four years, 0.23% of land area, which is more than double the land area for tree planting at 0.08% (Figure 4). Whereas England planted

0.06% of its land area with trees but also received SPHNs for an equivalent of half of that area at 0.03%. Scotland displays a less sizable application of SPHNs and no data was available for Northern Ireland. However, this does not include larch removal within the South West Scotland management zone, where SPHNs are not issued but larch is still likely being felled due to disease.

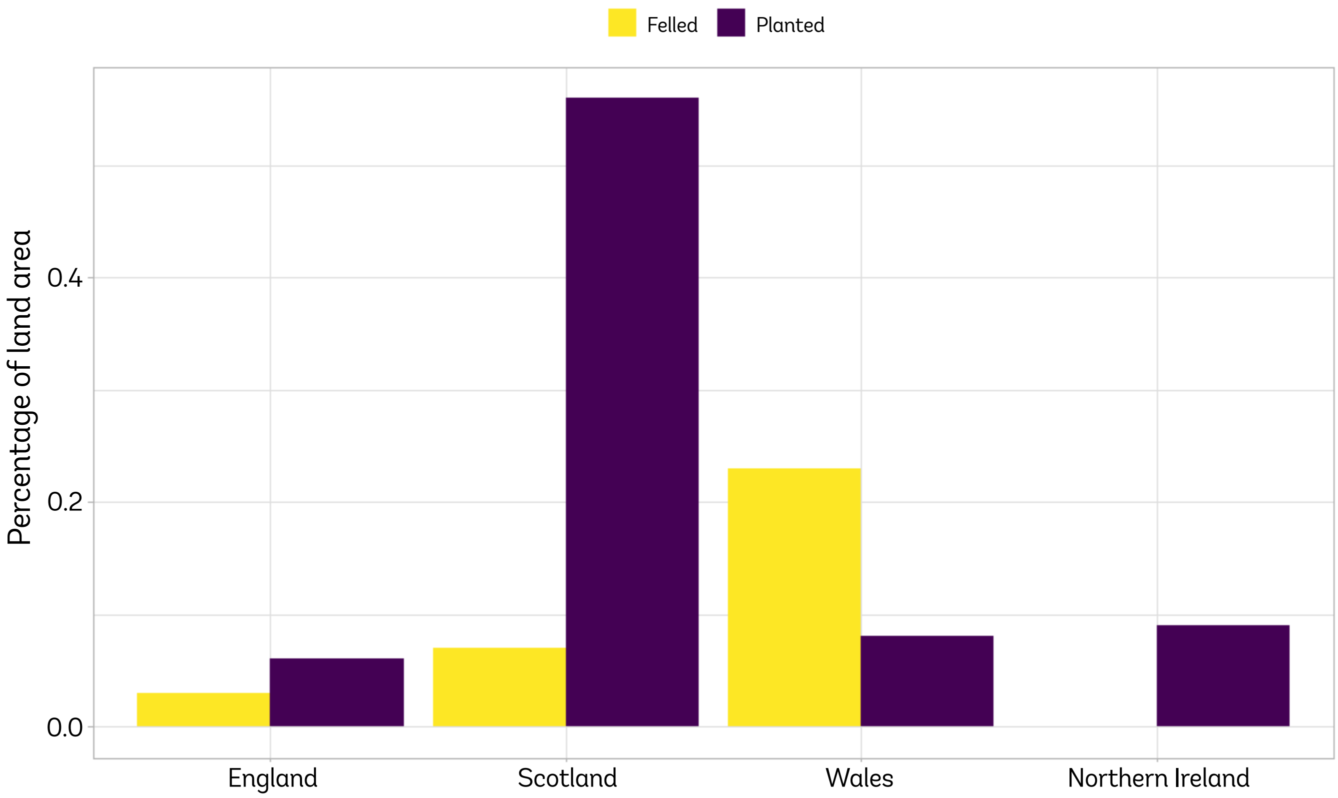

Figure 4. Percentage of land area which statutory plant health notices for ***Phytophthora ramorum*** were applied for each country in the UK, from April 2018 to April 2022, compared to percentage land area of tree planting. Although there is no data available, it is expected that the felled areas are also likely to be restocked (Maxwell, 2023).

Although SPHN felling of *P. ramorum* is significant, this is not representative of all the tree losses from introduced pests. For example, it is estimated that 12% of broadleaf woodlands are ash dominated: there are an estimated 185 million ash trees across the UK both inside and outside of woodlands, with a potential two billion further saplings and seedlings (Defra, 2019; Maxwell, 2023). As estimated mortality rates reach 70–85% for some woodlands, ash dieback is also causing large scale unrecorded losses of trees (Coker et al., 2019). It is unknown how many trees are lost each year to introduced pests, due to either management activities or from the pest itself.

Economic impacts

In 2023, a paper published by Eschen et al., provided an estimate of the annual cost of six introduced tree pests in the UK (Table 2). This includes money spent by government institutions and other stakeholders, not only on management for control, but also on research programmes (for example, biocontrol projects), costs of surveillance, yield losses, and loss of social and environmental value (Eschen et al., 2023). This demonstrates that for only six introduced tree pests, the annual cost to the UK is currently estimated to be £919.9 million (Eschen et al., 2023). The total cost of ash dieback over the next 100 years is predicted to be £15 billon (Hill et al., 2019).

Species	England	Scotland	Wales	NI	UK
Ash dieback (Hymenoscyphus fraxineus)	£556.4	£120.3	£125.9	£80.9	£883.5
Box tree moth caterpillar (Cydalima perspectalis)	£15.4	£0	£0	£0	£15.4
Green spruce aphid (Elatobium abietinum)	£1.4	£10.3	£2.4	£0.5	£14.5
Phytophthora ramorum	£0.7	£1.5	£1.9	£0	£4.2
Oak processionary moth (Thaumetopoea processionea)	£1.9	£0	£0	£0	£1.9
Great spruce bark beetle (Dendroctonus micans)	£0.1	£0.3	£0	£0	£0.4
Total (million)	£575.9	£132.4	£130.2	£81.4	£919.9

Table 2. Cost of managing tree pests and pathogens in the UK, including figures supplied by Government institutions and other stakeholders such as landowners. Cost figures include the cost of management, but also money spent on research, surveillance, yield losses and social and environmental value (£million) (Reproduced from Eschen et al., 2023).

Question five: What do we know about the future risk posed by further pest introductions?

Case study: Tree species and their associated biodiversity at risk

Ash and elm have experienced widespread losses due to an introduced pathogen. What else is at risk when we consider other tree species?

Oak

Number of oaks in the UK: estimated 219,000 hectares of oak-dominated woodland

Number of species it supports*: 2,300, of which 320 are entirely dependent on oak (Mitchell et al., 2019)

Current number of detected pests and pathogens: 14

Number of future threats: 35

Birch

Number of birch in the UK: estimated 236,000 hectares of birch-dominated woodland

Number of species it supports: over 520 insect species, with 110 entirely dependent (Mortimer et al., 2000)

Current number of detected pests and pathogens: 10

Number of future threats: 20

Scots pine

Number of Scots pine in the UK: estimated 208,000 hectares of Scots pine-dominated woodland

Number of species it supports: Initial results from the DiversiTree project suggests around 1,500 species are associated with Scots pine with over 200 entirely dependent (Pers. comms. Ruth Mitchell, 2024)

Current number of detected pests and pathogens: 18

Number of future threats: 60

*It is important to note that this is the number of species the given tree species has been shown to support. It doesn't mean woods with those species in will automatically support all the species listed as this depends on many factors e.g. connectivity of the landscape, age of tree, management of woodland etc"

The Plant Health Risk Register includes 339 pests which are absent from the UK but pose a risk to native tree species (Defra, 2024a). Of these, 10 are of high risk, and all these high-risk pests could be introduced through the movement of live trees (Table 3).

	High risk 60 and over	Medium risk 30-59	Low risk 0-29	Not yet assessed
Total number for our native tree species	10	106	199	24
Number that could move via live plant trade	10	97	143	N/A

Table 3. Number of pests on the UK's Plant Health Risk Register that have a native tree species listed as a host, for all pathways and for the live plant pathway only (Defra, 2024a). The risk rating is taken from the UK's Plant Health Risk Register's UK relative risk rating (mitigated). This considers likelihood of arrival, establishment and spread, the impact of the pest (including economic, environmental and social) and value at risk (Defra, 2024a). It also takes into account any current phytosanitary measures which mitigate the risk of the pest entering the UK.

Discussion

Introduced tree pests are having a large impact on trees and woods in the UK. The data demonstrates that a new serious pest of trees has been introduced on average every 1.1 years since 2000, and the UK plays host to 121 pests of our native tree species which are either introduced, or have uncertain origin (Figure 1). This impact is wide reaching, including the potential for every native tree species to play host to one of these pests. However, this is still likely an underestimate, as many plant pathogens, such as viruses and viroids, are still unknown to science (Wang and Zhou, 2016). Additionally, both the datasets used to determine this figure are incomplete, in part due to uncertainties around the presence and introduced status of many pests. Introduced pests can also spread rapidly to cover large areas and whole countries. Ash dieback, first recorded in 2012, is now found across much of England, Wales and Northern Ireland, and is spreading into the highlands of Scotland where ash populations are more limited (Figure 2). Oak processionary moth (*Thaumetopoea processionea*) is subject to an intensive control programme led by the Forestry Commission to slow its spread, however, it is still demonstrating a spread outwards from London. Additionally, an outbreak of this moth species was also found in Derbyshire in 2023 (Forestry Commission, 2023c). Between all the introductions that have occurred in the UK and the extent in which some of these pests have spread, it is becoming increasingly likely that an introduced tree pest will be present within each of our woodlands.

A significant factor behind the recent rise in pest introductions is the global movement of plants and plant-based goods (Boyd et al., 2013; Eyre et al., 2013; Potter and Urquhart, 2017; Roques, 2010; Woodward et al., 2022). The trade of live trees has been attributed to many pest and pathogen introductions, including oak processionary moth, sweet chestnut blight (*Cryphonectria parasitica*), and ash dieback (Liebhold et al., 2012; Potter and Urquhart, 2017; Smith et al., 2007). Despite this high biosecurity risk, the UK is still importing huge numbers of trees each year (Figure 3). The value of these imports is insignificant, £259 million in 2022, when compared to the annual cost of managing the introduced pests they threaten to bring, £919.9 million (Table 2) (Eschen et al., 2023). The UK does grow a substantial proportion of its planting stock domestically, an estimated 160 million trees annually (Forestry Commission, 2024b). Therefore, supporting this production is required to strengthen biosecurity and protect our trees and woods. The data also suggests that there is a recent trend towards importing larger trees that are more valuable. Future research should investigate whether this is a real change towards larger, more valuable trees, or a result of changing methodology. Importing mature trees presents a bigger biosecurity risk due to difficulty of inspection. Their longer growing time before import allows more interaction with pests, and large root balls are difficult to inspect.

Climate change is also often quoted as a driver behind increasing pest introductions, however this presents a complex picture. To date, no introductions have been evidenced to be caused solely by climate change and neither has it been proven to have aided any establishments. This is, however, predicted to change in the future as warmer weather and wetter winters create more conducive environments for some pests and increases susceptibility within trees through heightened stress (Harvell et al., 2002; Krokene 2015; La Porta et al., 2008; Pope et al., 2021; Simler-Williamson et al., 2019; Wainhouse and Inward, 2016; Watts et al., 2015). The complexity of climate and its impact on phenology, natural enemies, distribution of pests and hosts, and the impact of microclimates, does mean that these predictions might not present a complete picture (Simler-Williamson et al., 2019; Wainhouse and Inward, 2016).

The UK hosts significant diagnostic and outbreak response capabilities which have been successful in preventing outbreaks through intercepting pests on imported material (Table 3) and eradicating outbreaks, for example the Asian longhorn beetle outbreak in Kent in 2012 (Eyre and Barbrook, 2021). However, prevention is still the most effective tool. It has since been suggested that even high levels of intervention within the early stages of the Dutch elm disease epidemic would not have reduced the losses in the long term, demonstrating the importance of prevention to protect our landscape (Harwood et al., 2011). Work through the UK's Plant Health Risk Register is an important step within biosecurity procedures and allows mitigation measures to be implemented where a risk has been demonstrated (Defra, 2024a). However, the complexity of these introductions, predicting where they might occur from, especially when data availability is highly variable around the world, and the potential for novel interactions, presents challenges to understanding where and when the next introduction might occur (Burgess et al., 2015; Ghelardini et al., 2016; Gougherty and Davies, 2022; Santini et al., 2013; Wingfield et al., 2016). This is also difficult to address on a regulatory basis. World Trade Organisation (WTO) members must abide by the sanitary and phytosanitary agreement which requires members to allow imports unless there is an evidenced risk to human, animal or plant life or health – something which is difficult to do with so many unknowns (Gougherty and Davies, 2022; WTO, 2024). Examples of pathogens previously unknown to science until introduced into a new region on imported planting stock include red band needle blight (*Dothistroma septosporum*) and horse chestnut bleeding canker (*Pseudomonas syringae* pv *aesculi)* (Brasier, 2008). Additionally, taxonomic uncertainties can cause difficulty in preventative action. The causal agent suspected to be causing ash dieback was originally considered to be a native fungal species, *Hymenoscyphus albidus*. By the time it was accepted as a separate species, *H. fraxineus*, it had likely already been introduced to the UK (Baral, 2014).

The total number of trees lost to introduced pests and pathogens is unknown but SPHN data indicates that losses are likely large and widespread (Figure 4). The impact from these losses has many facets. This includes the impact on the biodiversity that uses the tree or is present within the tree's ecosystem, the loss of ecosystem services the trees supply (such as carbon storage), the economic cost and the loss of heritage, wellbeing and our enjoyment value of woodlands (Boyd et al., 2013; Eschen et al., 2023). Typically, landowners are burdened with the management costs. That may be managing the trees due to health and safety concerns, which is commonly performed for ash dieback, managing the pest itself due to its human health impacts, such as oak processionary moth which can cause rashes and allergic reactions in people and animals, or managing due to statutory requirements, such as for *P. ramorum*. The economic costs are much wider than the cost of management alone and include research, surveillance activity, yield losses and loss of social and environmental values (Eschen et al., 2023). Six introduced pests alone cost an estimated £919.9 million annually and this figure does not include other costly pests that are under active control, including sweet chestnut blight, *Phytophthora pluvialis*, and *Ips typographus* (Eschen et al., 2023).

Conclusion

The difficulty in predicting the future for tree pests and pathogens is complex and includes many unknowns including how pests will be impacted by future climates, what hosts or vectors they might use in novel environments and the influence of international trade of live plants as supply and demand changes. The increasing number of pests creates a cumulative effect which weakens resilience within our treed landscapes. Existing biosecurity procedures in the UK provide an important and essential step towards safeguarding our trees and woods, however, another vital prevention method is to reduce the importation of trees. One of the key drivers of these introduced pests can be addressed through growing trees domestically in the UK. Purchasing trees which have been grown in the UK for their entire life is the only way to ensure that the tree is not introducing a new pest to the UK.

What needs to happen?

Tree buyers

- Specify UK grown when purchasing trees. The Woodland Trust's UK and Ireland Sourced and Grown scheme (UKISG) provides the confidence that trees have been grown in the UK for their entire life. Ask for UKISG trees from these tree nurseries - Where Do Our Trees Come From? - Woodland Trust.

- Consider using contract growing to provide tree nurseries with enough notice to grow the tree species and provenances you need in the UK, without the need to import.
- Where there are shortfalls in UK grown tree supply due to a lack of seed supply, consider identifying and setting up seed stands on your land, or working with local landowners to do so, to help boost seed supply in the UK.
- When procuring larger mature trees for landscaping schemes or urban planting, use a 'UK grown' first approach and provide as much notice as possible in tree procurement activities.
- Improve biosecurity when importing large trees through innovation and infrastructure. Large tree imports pose the highest biosecurity risk due to their size and large root balls creating difficulty in inspection, and their longer life increasing the chances of pests and pathogens being present. Currently, border control points are not well equipped to inspect large and mature trees and shrubs under cover. Governments should work with port authorities to support investment in biosecure facilities to allow thorough inspection of these trees before they move into the wider environment.
- Implement a 'UK grown' kitemark for UK grown plants and trees to allow consumers to make a conscious decision and support our UK growers. As recommended by the House of Lords horticultural sector committee's report *Sowing the seeds: A blooming English horticultural sector.*

Key evidence gaps:

- Understanding the scale of losses of trees from introduced pests, and how these losses impact biodiversity and ecosystem services.
- The impact of climate change on the risk of pest introductions, and what, if any, mitigation measures can be implemented.
- Does the changing trade data represent a consumer change that is increasing imports of larger, more valuable trees? If so, how can we give tree nurseries the confidence they need to grow mature and semi-mature tree stock in the UK?
- Detail of what a world-class biosecurity system for tree protection should look like – how much would this cost and would it be less than the management of pests per year?

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Tree supply

UKISG

The Woodland Trust ensures that every tree it sells, plants, or funds is sourced and grown in the UK or Ireland. Our voluntary UK and Ireland Sourced and Grown (UKISG) scheme, is the most reliable source of these trees, it is an externally audited standard that ensures good biosecurity practices and the trees' origin thereby reducing the likelihood of disease importation. The Trust procures around six million UKISG trees annually. Established in 2014, the scheme provides confidence that the tree has been grown in the UK for its entire life using UK sourced propagation material. There are currently 51 tree nurseries in the UK who are UKISG audited and assured. This includes large forestry nurseries, small nurseries, community tree nurseries, and amenity tree nurseries. UKISG can be applied to any species of tree, at any size.

Plant Healthy Certification Scheme

The Plant Health Alliance, which consists of organisations from horticulture, forestry and land management, including the Woodland Trust and government departments, published the Plant Health Management Standard in 2019. This voluntary standard aims to boost biosecurity practises across the horticultural and arboricultural industries, including within nurseries, professionals, and retailers. Plant Healthy is a certification scheme for those who are audited and meet the Plant Health Management Standard. The main purpose of the scheme is to control threats to the wider horticultural trade from notifiable pests. The participating organisation must complete a comprehensive risk assessment to prepare for their Plant Healthy audit to ensure transmission pathways for pests are recorded and risk assessed, and any control measures are in place. This scheme is a valuable tool to help improve biosecurity standards across the sector. Plant Healthy and UKISG work together to provide an option for tree nurseries to join both schemes in one audit.

The impact of deer on UK woodland resilience

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Summary

Six deer species are present in the wild in Great Britain (two native species, one naturalised species and three invasive species). Only four species are present in Northern Ireland (one native, one naturalised and two invasive). Sustainable levels of deer perform vital ecological functions which maintain ecological dynamism. Increases in numbers and ranges of all species present in the United Kingdom now pose a threat to woodland resilience in a changing world.

Resilience is defined as the ability of an ecosystem function to resist or recover rapidly from environmental perturbations. There are no empirical studies explicitly testing the response of woodland resilience to changes in deer pressure in the UK, although there is sufficient evidence demonstrating the effect of different deer densities on woodland complexity and structure, which can be used to infer an effect on overall resilience. Woodland creation and restoration targets are being challenged by deer impacts and carbon sequestration targets via tree growth will also be affected.

Here we review the available literature on the effects of native and non-native deer on native woodland resilience in the UK. Evidence gathered from across the Woodland Trust estate is used to add context and illustrate the issue of deer impacts further.

High deer densities have been shown to negatively affect several ecological components which contribute to woodland condition across the UK. Deer have been shown to negatively affect tree regeneration and tree growth response to climatic factors such as temperature, woodland vegetation structure, ground flora species richness, abundance of birds, small mammals and invertebrates, nutrient cycles and soil biological properties. However, at lower densities, deer have been shown to benefit many of the same ecological components, highlighting their importance in woodland ecosystems at sustainable densities.

On Woodland Trust sites high numbers of deer are being recorded, and generally high impact scores on woodland vegetation. Without significant investment and intervention across the sector, woodland structure and ecology is, and will be, adversely impacted.

Management of deer at landscape scale is needed to effectively reduce their impact on native woodlands and ensure impact levels remain low following management. Recent primary literature and grey literature, as well as population models, provide guidance on the scale needed to effectively manage different deer species, and the number, sex and age class which require culling to maximise efficacy. Examples are provided of current landscape partnerships which provide case evidence which can be used to

unlock funding and upscale deer management across the UK.

Barriers and research gaps are also identified. Development of survey methodologies capitalising on new technology is required to provide better estimates of the current deer population and enable the development of country and regional level population and impact assessments. The development of a range of population density estimates above which ecological impacts are likely, drawing on species, regional and climate data is recommended. Furthermore, additional studies testing the scale at which management is likely to be effective are needed.

Introduction

There are six species of deer which live in the wild in the UK. Two of these species, red deer (*Cervus elaphus*) and roe deer (*Capreolus capreolus*) are native (although current populations are largely descended from reintroduced populations), while the fallow deer (*Dama dama*), muntjac (*Muntiacus reevesi)*, Sika deer (*Cervus nippon*) and Chinese water deer (*Hydropotes inermis*) are non-native introduced species. While muntjac, Sika deer and Chinese water deer are relatively recent introductions (all introduced in the 19th and 20th centuries) and are considered invasive species, fallow deer is a long-established species first introduced by the Romans and reintroduced in the 11th century and is considered a naturalised species (British Deer Society, 2020a).

Increasing deer populations and problems with overabundance

Deer (red and roe) have been present in British woodlands to varying degrees for thousands of years (although archaeological records show very limited evidence of these species during the last millennia) and play a vital role in maintaining woodland dynamism via important ecological functions including herbivory, seed dispersal and nutrient cycling (explored further in this chapter).

Following historical declines due to forest clearance and over-hunting, deer populations have now been increasing in the UK for approximately 200 years (Fuller et al., 2001, Dolman et al., 2010). However, high density deer populations are widely acknowledged to detrimentally impact woodland ecosystems, for example, by subjecting trees to direct damage as a result of herbivory, bark stripping or fraying (Gill, 1992) or indirectly altering woodland structure. Rising deer numbers also has important implications for human-wildlife conflict such as traffic collisions and transmission of Lyme disease (British Deer Society, 2020b). The Woodland Trust has received reports of deer attacking dogs where pressure is particularly high. Issues relating to human-wildlife conflict will not be discussed further as they are out of scope of this review.

There are several factors which have influenced recent deer increases, including higher birth rates (Moyes et al., 2011), recent woodland expansion and land abandonment, agricultural changes leading to increased prevalence of winter cereals and therefore fewer losses, a reduction in livestock husbandry in lowland woodlands, increased control and regulation of deer hunting, a warming climate and elimination of large predators (Fuller and Gill, 2001). A lack of management partly due to limited sector capacity and poor venison prices also pose challenges.

Intentional, business-driven approaches where supplementary feeding is provided by upland sporting estates is also likely to have contributed to the observed increases in numbers by providing more favourable conditions for deer populations, as is legislative protection for deer species. While it is likely that these factors have likely had an interactive effect on rising deer populations, there is little evidence available which explicitly aims to determine the relative effect of each, and it is unlikely that all factors have had an equal effect. For example, the lack of large predators in the UK is often cited as a primary driver of rising deer populations and it is possible that this lack of predators is disproportionately responsible for recent population growth. Indeed, the UK is almost unique in Europe in having been without such predators for several hundred years. However, further study aiming to understand the relative role of each of the historical drivers outlined above would be beneficial here to allow for research and management prioritisation.

In 2010, the deer population in Britain was estimated at 1.5 million (Dolman et al., 2010). More recently, the UK deer population is commonly cited as being two million, although there is much uncertainty surrounding the accuracy of this estimate (British Deer Society, 2023a). While in many cases deer population size alone is likely to be insufficient to enable a prediction of the damage expected to be caused to native woodland, it is clear that in many areas of the UK expanding deer populations significantly exceed those at which negative effects of deer are anticipated (Putman et al., 2011).

In a changing world, facing dual threats from the biodiversity and climate crises, it is more important than ever that woodlands and landscapes in the UK are resilient to external threats. If deer populations are so high that they affect ecological processes such that woodlands are unable to sufficiently withstand and adapt to change, the future of woodland itself in the UK may be threatened with long-term consequences for threatened woodland biodiversity.

Defining resilience

Ecological resilience can be defined as 'the degree to which an ecosystem function can resist or recover rapidly from environmental perturbations' (Oliver et al., 2015), and is understood as an emergent property of ecological complexity, in turn described as 'number of components in a system and the number of connections among them' (Bullock et al., 2021). It is in the context of these definitions that resilience will be discussed throughout the review. In addition, habitat condition can be linked to habitat complexity. Condition is discussed in detail in the condition section of this report.

In the context of woodlands in the UK, the components of woodland resilience comprise the diversity of tree species and other species in the woodland ecosystem, the genetic variation within species, the wider regional, genetic pool of species and ecosystems, and the extent, condition and character of the surrounding landscape (Spencer, 2018). Factors such as landscape connectivity and woodland structure also have the potential to affect woodland resilience via their effects on the dispersal of species (including native, naturalised and invasive species, as well as diseases) and the provision of suitable niche space, respectively.

Woodland surveys in Scotland provide evidence of the potential effect of deer pressure on woodland extent and condition. Results from the native woodland survey of Scotland (Forestry Commission Scotland, 2024) indicate that over 10% of Scotland's ancient woodland area has been lost over the last 50 years, with the most likely drivers of change cited as being herbivore pressure and poor regeneration capacity of older trees. Similarly, Trees for Life's comprehensive Caledonian pinewood survey (Trees for Life, 2023) revealed that 23% of said pinewoods are in critical condition, with inappropriate levels of deer browsing again cited as a major driver.

While there are no experimental studies looking explicitly at the effect of deer on woodland resilience as a whole, there are numerous studies investigating the effect of deer on individual components of woodland condition.

As the UK's largest woodland conservation charity, the Woodland Trust invests considerable time, effort and money to help understand the threat and mitigate the impact of high numbers of wild deer. This has included the use of new technologies and improving survey methods. Drones equipped with thermal and optical cameras allow for accurate counts of deer to be undertaken. Herbivore Impact Assessment surveys allow for impact scores to be collected and analysed. Cull data allows for a considered and appropriate response towards reaching conservation objectives. The Woodland Trust estate covers more than 33,000 hectares or 330 km^2 across all four UK nations.

Here we review the available literature on the effects of native and non-native deer on native woodland resilience in the UK. Evidence gathered from across the Woodland Trust estate is used to add context and illustrate the issue of deer impacts further. The aim is ultimately to better inform guidance on how woodland resilience in the UK can be maximised, and further understand how deer populations and deer management may affect this aim.

Key questions identified prior to beginning the review which will be answered below are:

- What impacts do deer have on woodlands in the UK?
- How do deer affect woodland resilience, complexity and condition?
- How do deer impact the ability of woodland managers to achieve management objectives?
- What is being done to tackle the deer problem?
- What affects our ability to achieve successful management?

Methodology

A two-pronged approach was taken. A literature review was conducted, and Woodland Trust deer and herbivore impact monitoring data was analysed. Results were then synthesised to produce a more in-depth picture of the impact of deer on UK woodland resilience.

Literature review

The review draws on academic and grey literature and was carried out using guidance from Collaboration for Conservation Evidence (CCE, 2022). Google Scholar was used as the primary search engine for academic literature. Websites of relevant organisations (e.g, Applied Ecology Resources, CEH, BES, British Deer Society, The Deer Initiative, Forestry Commission, Forest Research) were utilised to search for grey literature and professional contacts were also utilised. Reference lists of included papers were also checked.

Evidence from the UK was prioritised, however evidence referring to deer (prioritising those species which are found in the UK) impacts in temperate regions outside the UK was also utilised where relevant. Search strings included the terms 'deer', 'wood*', 'tree*', 'United Kingdom' or 'Brit*' and 'resilience' or 'complexity' or 'condition'. Additionally, individual components which contribute to resilience, complexity and condition were used. For example, individual components which contribute to woodland condition as defined by the NFI such as 'regen*' or 'herb*' were used. In order to ensure that potentially relevant resources which were not returned using the criteria above were not missed, a simplified search string was also used which included only the terms 'deer', 'wood*' and 'resilience' or 'complexity' or 'condition'.

Analysis of Woodland Trust data

Three relevant datasets were available for analysis. Herbivore Impact Assessments (HIAs), Thermal Imagery Census data (TIC), and cull data. Analysis is restricted to the 2023-24 deer management season (July 2023 – May 2024) due to data availability and completeness. No TIC data was available from Northern Ireland and no HIA scores were available from Scotland during this period. The Woodland Trust's estate is sub divided by country and region (CAR) with Scotland, Wales and Northern Ireland as countries, and then England is divided into four regions: North, Central, South West and South East.

- Thermal Imagery Census (TIC) data was collected by flying drones with thermal and optical detectors over a woodland site to ensure as complete coverage as possible. Data is collected on deer numbers, species, sex and age (where possible). Photos and/or video footage is collected for records and to aid further identification. Surveys are generally flown during the late autumn, winter and spring months when the lack of leaves on broadleaf trees gives better visual clarity, and the cooler temperatures allow for better thermal detection. Deer densities were calculated as the number of deer seen in a survey divided by the total area of the site surveyed. There was no adjustment for areas that were not surveyed as this information is not routinely recorded. In total, data for 71 sites was available for the 2023-24 season, although there was none from Northern Ireland.
- Cull data comprises deer species ID and location, and in most cases sex and age. Cull targets are established based on a range of criteria and site objectives. Eradication is not always the target, which means deer counts from cull data do not accurately reflect deer density on site. However, cull data does provide some indication of species present and local populations. In total, cull data from 84 sites was available for the 2023-24 season.
- The abbreviated Herbivore Impact Assessment (HIA Lite) was used to assess deer impact on woodland vegetation (NatureScot, 2021). Multiple survey stops are visited within the woodland area to provide a representative assessment of the habitat. At each stop three different vegetation classes are assessed for herbivore impacts. Each variable is scored between 0 (no impact) to five (very high impact). Scores are averaged to produce an overall impact score and assigned a category (no impact, low impact, medium impact, high impact and very high impact). In

total HIA data was available from 134 sites for the 2023-24 season. There were no HIA scores available from Scotland.

- Data is collected on an opportunistic basis to inform management decisions rather than as part of a systematic sampling strategy and is therefore not necessarily representative of the Woodland Trust estate or the national picture. However, due to the paucity of national-scale deer data, it represents a useful dataset to add context and illustrate the issue of deer impacts.

Results and discussion

UK deer species composition and distribution

Native species

Red deer, the largest species in the UK, are most common in Scotland and are also common in the South West, East Anglia and areas of northern England, with populations scattered throughout the rest of the country. Found predominantly in woodland and forested habitats, they also utilise more open moorland habitat. The species feeds primarily on grasses and dwarf shrubs however woody browse, such as tree shoots, is also taken. Roe deer are territorial and strongly associated with woodland habitats. They are particularly associated with woodland edges and browse on highly nutritious plants. They are also frequently found in scrub and hedgerows. They are significantly smaller than red deer, and are abundant across the UK, although they are not found in Northern Ireland (British Deer Society, 2020a).

Both native species were observed in Scotland and England in the Woodland Trust 2023-24 GB deer survey season. Although the density of red deer that was observed is low, this only reflects what is happening at the Woodland Trust sites where TICs were carried out. This does not represent the more general picture across both the Woodland Trust estate, and particularly Scotland, where red deer densities are often known to be high (British Deer Society, 2020a). Roe deer were observed in Wales but not red deer, potentially reflecting their relative scarcity in Wales currently. In Northern England most sites were dominated by roe. See Figure 1 for composition of deer species observed from a TIC and Figure 2 for species recorded at cull in the 2023-24 season.

Non-native species

Fallow deer are a medium to large-sized deer found commonly in mature broadleaf woodland with understories, open coniferous woodland and open agricultural land. Originating from escapee populations from deer parks, they are now widespread in England and Wales but less common in Scotland and Northern Ireland. The species is locally abundant and increasing, and although they preferentially graze grasses, young tree shoots and dwarf shrubs are also taken. Due to their herding behaviour fallow deer can reach very high local densities and range across sites at a landscape scale (British Deer Society, 2020a). However, herds are also known to occasionally become strongly hefted to a specific woodland site. Fallow deer were very common in the South East and South West of England and in Wales during the Woodland Trust 2023-24 GB deer survey season (see Figures 1 and 2).

One of the smaller species in the UK, muntjac are now widespread, as observed during the Woodland Trust 2023-24 GB deer survey season (see Figures 1 and 2). Muntjac are present, but to a lesser extent, in Wales while small populations are also present in Northern Ireland. Deliberate releases and escapes from deer parks have established populations in the wild, and the species is now increasing in number and range. Muntjac are a territorial, browsing species and favour deciduous and coniferous woodlands with diverse understories. They are also found in scrubby habitats. The species is listed on Schedule 9 of the Wildlife and Countryside Act 1981 (as amended) as an invasive species, and since 2019 the species has been listed as an invasive species on the Invasive Alien Species (Enforcement and Permitting) Order 2019 (British Deer Society, 2020a).

Sika deer are another medium to large sized deer, and through deliberate release or escape have now colonised much of the UK. Sika deer are known to hybridise with native red deer leading to loss of native red deer genetic identity. Whilst sika have been recorded during previous surveys, no individuals were detected in the Woodland Trust 2023-24 GB deer survey season, suggesting they were missed or not present at the time of the survey. Populations of sika deer in Scotland are widespread and expanding from west to east. Sika distribution is patchier in England with geographical bands of abundance present in the north and south of the country, and they are also more commonly recorded in the west of Northern Ireland (where they are a significant conservation problem) than in the east. They prefer coniferous woodlands and acid heaths, and while they graze primarily on grass and dwarf shrubs, coniferous and broadleaved shoots may also be taken in small quantities (British Deer Society, 2020). The species is listed on Schedule 9 of the Wildlife and Countryside Act 1981 (as amended).

Chinese water deer are found predominantly in eastern England, although scattered records exist throughout England. Only nine individuals were culled in the Woodland Trust 2023-24 GB deer survey season (see Figure 2). They prefer habitats including reed beds, river shores and open fields, along with woodlands. They are selective feeders and particularly like nutritious herbs, although woody browse may be taken if food is limited (British Deer Society, 2020a). The species is also listed on Schedule 9 of the Wildlife and Countryside Act 1981 (as amended).

Deer observed on the Woodland Trust Estate (2023-24)

Of the six species of deer in the UK, four were observed by TIC in the Woodland Trust 2023-24 GB deer survey season: red, roe, fallow and muntjac. Figure 1 shows a map of the sites where deer were observed and the breakdown of species at each site. These four species were also extremely common in cull data (see Figure 2), with the addition of two other non-native species not detected by TIC. Sika (nine in Scotland) and Chinese water deer (nine in Central England) were detected via cull data, albeit in small numbers.

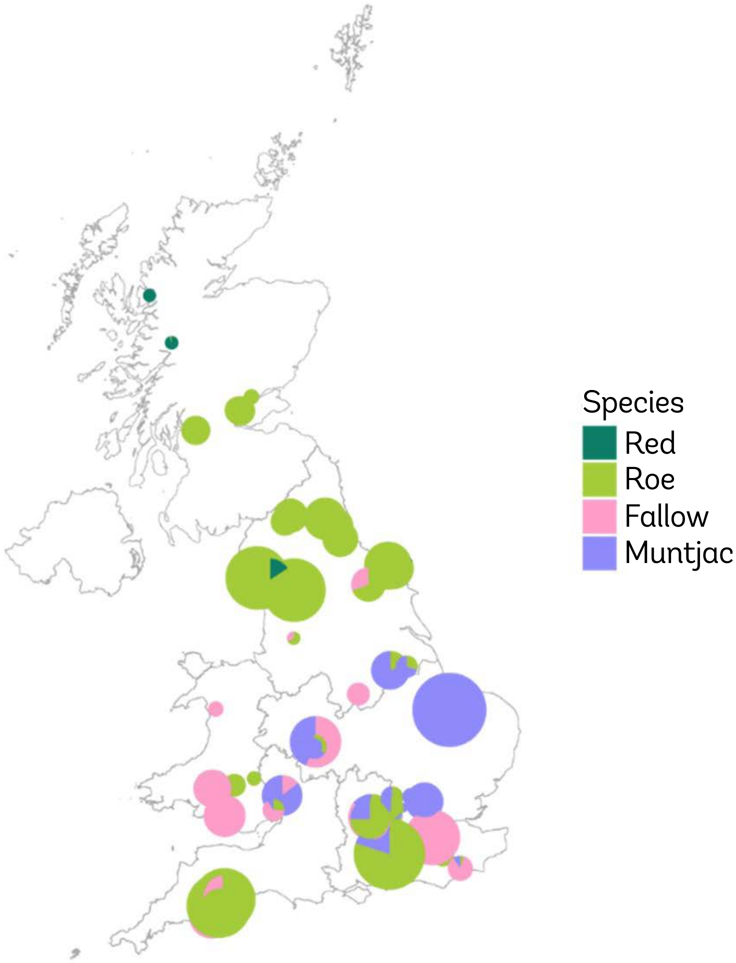


Figure 1. Species composition of deer observed from TIC in 2023-24. Area of dot is proportional to deer density at that site ranging from 2/km^2 to 139/km^2. Some dots are overlapping but general patterns can be observed. No TICs were carried out in Northern Ireland in 2024. Sites chose to carry out a TIC based on a perceived requirement for a wildlife management plan and so do not represent a systematic or random sample of Woodland Trust sites.

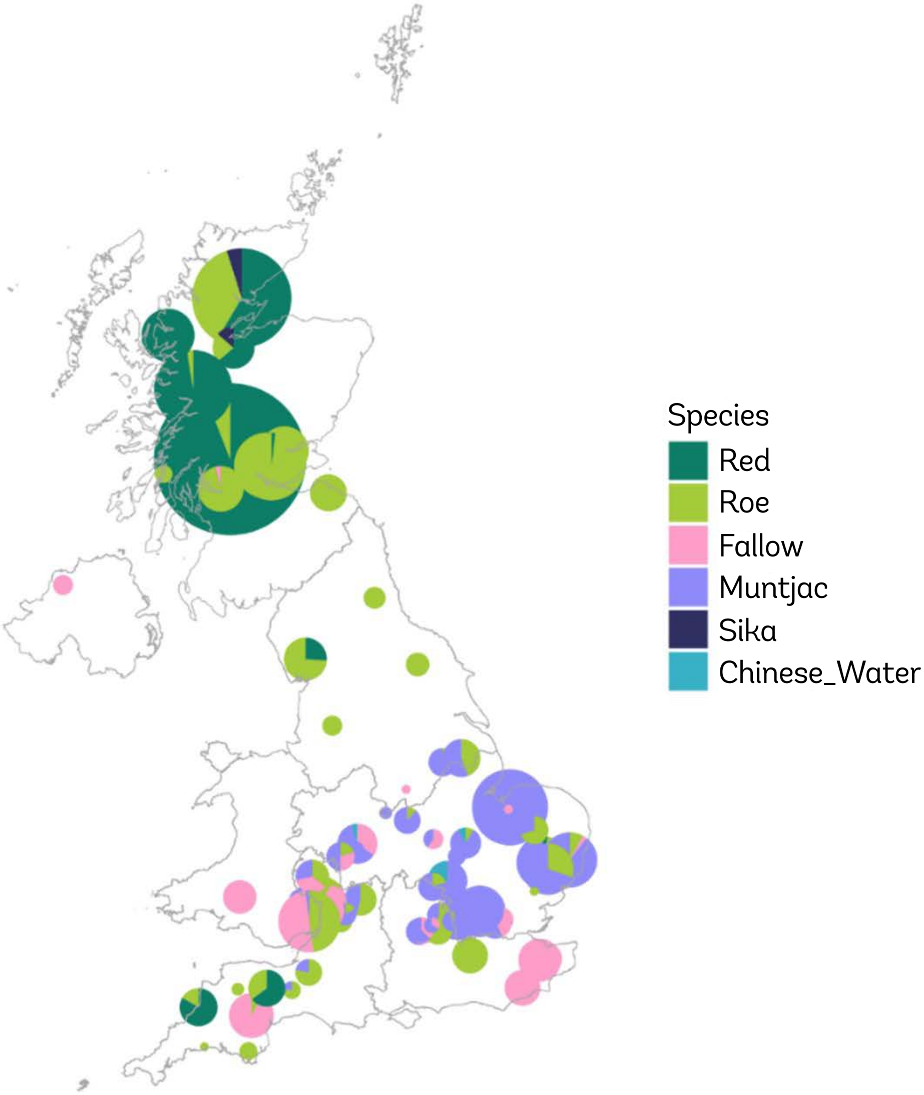
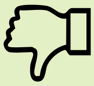


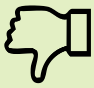
Figure 2: Species composition at sites where cull data was returned for the 2023-2024 season. Area of dot represents total number of deer culled at that site during the season ranging from one to 288 deer. Overlapping dots are due to sites being close together. Sites shown are those that culled deer and returned cull records so do not represent a systematic or random sample of Woodland Trust sites.

Ecological impacts

Table 1 summarises the ecological impacts of deer in the UK on tree regeneration, vegetation structure, ground flora, fauna, nutrient cycles, seed dispersal and climate. These categories are expanded upon in the text.

Table 1. Summary of the effects of high deer populations on native woodlands

Impact/ outcome	Positive or negative impact*	Scale of impact*	Strength of evidence	Key message
Tree regeneration		High	High	There is strong evidence that high deer densities can negatively affect tree regeneration, and that reducing deer populations to more sustainable levels can allow large-scale woodland regeneration. There is also evidence that in lower densities deer can facilitate tree regeneration via their effects on vegetation structure.
Vegetation structure		High	High	There are several examples in the literature of high deer densities simplifying woodland structure, leading to a reduced ground and shrub layer, with important implications for woodland fauna.
Ground flora	?	High	High	Deer grazing has the potential to significantly affect ground flora communities and lead to dominance of unpalatable, grazing tolerant and/ or nitrophilous species. However, deer may increase species richness in areas where they reduce the cover of dominant species, such as bramble. They may also improve conditions for lower plants.
Fauna	?	High	Moderate	Via their effect on woodland structure, deer can negatively affect several faunal taxa. Woodland birds and small mammals requiring dense understories are likely to be particularly affected. Deer may also benefit several species by opening dense understories and creating glades and rides.

Impact/ outcome	Positive or negative impact*	Scale of impact*	Strength of evidence	Key message
Nutrient cycles	?	Unknown	Low	Selective deer browsing has the potential to negatively affect nutrient cycling and soil biological properties via an effect on leaf litter quality and a limiting effect on tree regeneration. Urination and defecation return labile nutrients to soil, which can be considered a positive effect, although this may be offset by the effects outlined above, and lead to eutrophication.
Seed dispersal		Unknown	Low	Deer distribute a range of plant species. This has important implications for woodland dynamism and can facilitate movement of plant species from areas of mature woodland to younger, more open woodland. Deer may also have a negative effect on invasive plant dispersal.
Climate		High	Low	Deer mediated changes in vegetation structure may benefit certain tree species in warming climatic conditions due to reduced plant-plant competition, although significant browsing can reduce sapling resilience in the same conditions. Deer may also affect carbon sequestration goals and increase peatland erosion.

*Where evidence is lacking or weak, impact assessment is based on expert judgement and ecological principles.

Tree regeneration and deer density

A key component of woodland resilience is the capacity for the woodland to regenerate naturally. This provides a mechanism for adaption to changing environmental conditions (Cavers and Cottrell, 2015). Therefore, factors preventing the natural regeneration of tree seedlings and saplings are likely to limit resilience in the long term. There are various scientific studies which provide examples of deer limiting natural regeneration. For example, Gill and Morgan (2010) found that naturally regenerating seedling density is negatively correlated with deer population density due to browsing of seedlings and saplings. Furthermore, preferential browsing by deer can favour regeneration of unpalatable species and limit the regeneration of more palatable species, changing the composition of woodland communities over time and further reducing complexity and resilience. Deer densities of more than 14/km^2, indicates regeneration is likely to be limited (Gill and Morgan,

2010) without other interventions such as fencing.

The 14/km^2 deer density threshold differs in different habitats and contexts, depending on landscape factors and the deer species present. Indeed seedling density was also negatively correlated with the proportion of larger deer species recorded, which in this case was mainly fallow (Gill and Morgan, 2010). For example, in native Scottish pinewoods successful woodland regeneration has been shown to require red deer densities below 4/km2 (Gullet et al., 2023). However, the most palatable tree species may require periods of lower deer densities to successfully regenerate (Gullet et al., 2023). Further evidence that sustained deer culling at the landscape scale can significantly reduce deer numbers and increase woodland regeneration rates comes from the Mar Lodge estate, also in the Cairngorms, where similar results have been reported (Rao, 2017).

In a long-term experiment in South West Ireland, (Perrin et al., 2006), sika deer exclusion was found to strongly affect tree regeneration in woods dominated by both yew (*Taxus baccata*) and oak (*Quercus petrea*). Very few yew saplings or seedlings were recorded inside or outside of the exclosures during the study period, highlighting the need for an understanding of species shade tolerance and the interaction between deer browsing and light regimes when attempting to understand lack of regeneration (Perrin et al., 2006). In this context, deer grazing may alter long-term woodland species diversity if shade intolerant species are grazed or browsed prior to canopy closure. It has also been suggested that without disturbance by large deer (such as red deer), tree regeneration can be stalled by other factors such as a lack of bare ground suitable for germination and increased rhododendron density (Scott et al., 2000; Perrin et al.,2006). This indicates that sustainable deer populations are part of a well-functioning and resilient ecosystem with appropriate levels of tree regeneration, but that densities in the UK are not close to this level currently.

At unsustainably high numbers, all deer species can have a negative impact on tree regeneration. The UK has a number of non-native or naturalised species present. However, it is not possible to make broad statements regarding the relative severity of impact of either native or non-native deer species. The impact will vary based on the community of species present due to different ecological niches, traits and behaviours. Also, each woodland site is unique, meaning there will be variability in any given site's ability to tolerate high deer impacts.

Comprehensive data across the four countries of the UK on deer densities; overall and by species, is currently not available, therefore it is not possible to accurately predict the true scale of the impact of deer herbivory on UK woodlands.

We use the Woodland Trust 2023-24 survey season data from the TICs and HIAs to provide indications of deer densities and herbivore impacts, noting that the sites where data are available reflect the need for data to inform wildlife management plans rather than an overarching systematic or random sample of Woodland Trust sites. As such, the cull, TIC and HIA data is not necessarily representative of the Woodland Trust estate or the national picture. However, due to the paucity of national-scale deer data, it represents a useful dataset to add context and illustrate the issue of deer impacts.

Whilst site-scale deer density assessments are clearly necessary to inform management, coordinated, cross-sector and strategic monitoring at the landscape scale is needed to enable more accurate prediction of regional and national impacts.

Deer densities varied hugely between Woodland Trust sites, ranging from zero to 139/km². This high level of between-site variation is illustrated in Figure 3. There were 48 sites (68% of sites) with a density of over 14 deer/km² and 36 (50%) with a density of over 24/km². Some of the small sites had very high densities – five sites had a density of over 100 deer per km².

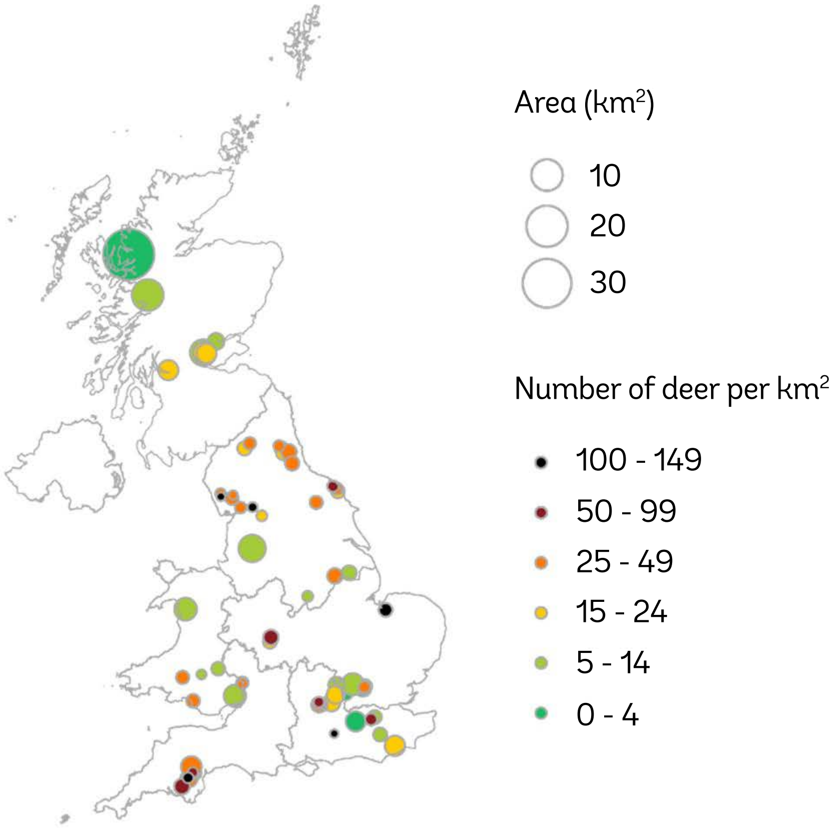


Figure 3. Deer density per km² at the 71 sites surveyed using TIC in the 2023-24 season. Colour indicates deer density. Area of dot indicates area of site. Some sites may be (partially) hidden due to many being surveyed in the same area. Sites chose to carry out TICs based on a requirement for a wildlife management plan and so do not represent a systematic or random sample of Woodland Trust sites.

An overall HIA category of medium, high or very high impact was observed on 63% of the 134 sites surveyed during the Woodland Trust's 2023-24 GB deer survey season (note: no HIA from Scotland were available for inclusion). See Figure 4. This indicates material damage to the woodland vegetation surveyed on site, including tree seedlings and saplings. There were 44 sites (38 in England, six in Wales) where both HIA scores and deer density estimates from TICs were available. Of these there were 32 sites with >14 deer/km². All but one of these had an HIA score indicating at least a category of medium impact on the vegetation present, but often higher (11 with high impact and three with very high impact). Sites which were classified as very high impact had an average deer density of 63 deer/km^2 (n=3). Of the 12 sites with ≤14 deer/km^2 six were categorised as medium impact and two with high impact.

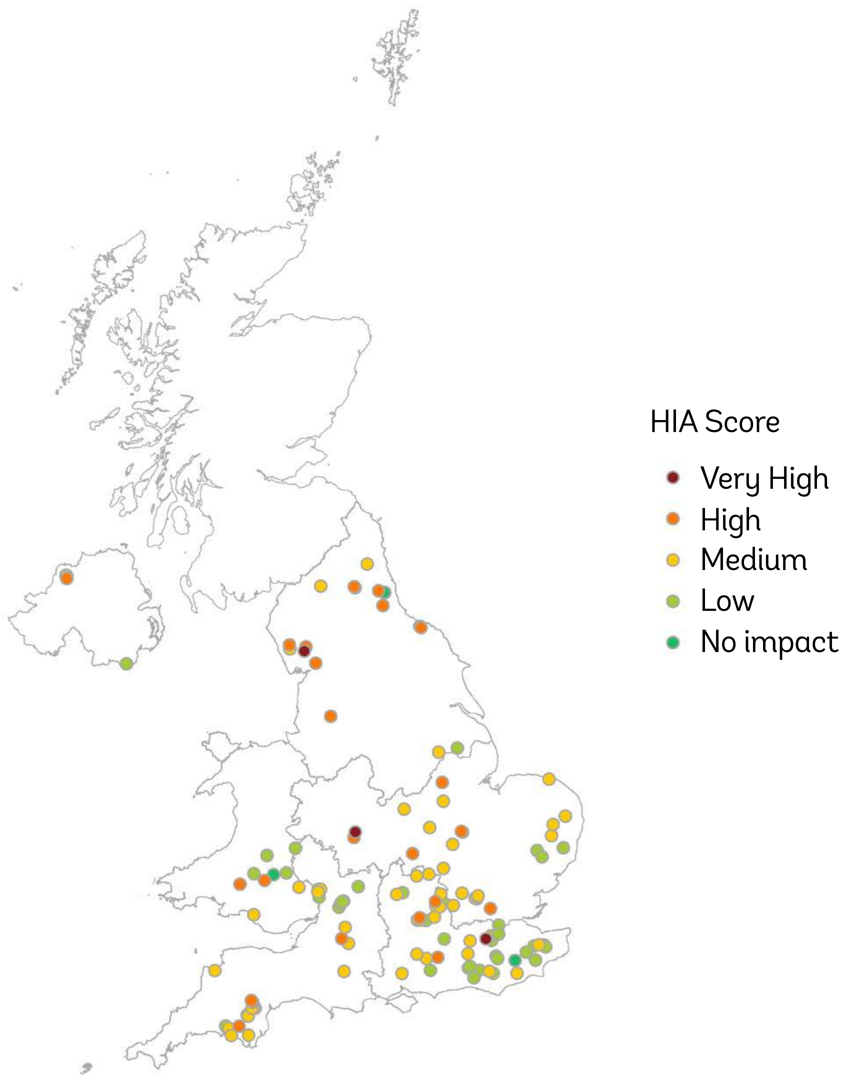


Figure 4. Map showing sites where HIAs were carried out in 2023-24 season. Colour indicates HIA score. Note: no data was available from Scotland at the time of writing and sites were not selected for assessment in a systematic or randomised way.

Many factors influence drone operation and detectability of deer via TIC: time and season of survey, visitor or local sensitivity of site, weather, woodland type and density, and understorey vegetation. Due to this, and the free-ranging nature of deer beyond the boundary of a site, the number of deer recorded is only ever a snapshot and may not reflect the number that impact a site. For example, at Fingle Woods in Dartmoor, Devon, many fallow deer were observed outside the site, which is not reflected in the count (see Figure 5), but these deer may at other times of the day be present on, and impact the vegetation of, the site. Crucially, this highlights the importance of landscape-scale deer management plans and efforts, with cooperation between neighbouring landowners to tackle deer impacts. This is not always easy with potentially competing objectives and ideologies.

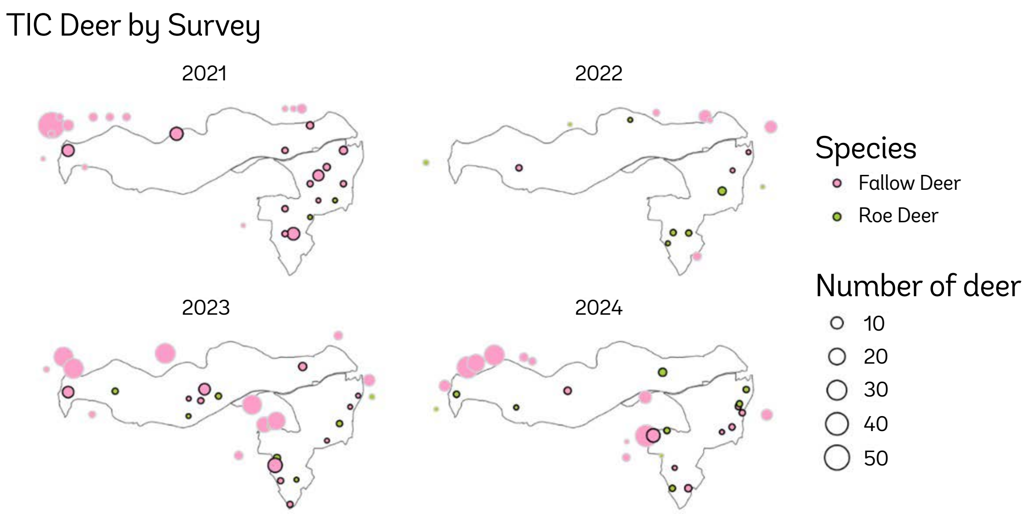


Figure 5. Maps of Fingle Woods showing location of groups of deer observed in the TIC in each of the 2021 – 2024 seasons. Colour of dot indicates species; area of dot shows size of group and border colour shows whether the group was observed inside (black) or outside within 300m (light grey) of the site.

Vegetation structure

A component of woodland condition which is likely to enhance resilience is varied vegetation structure, which is now an uncommon sight in woodlands across much of the UK. Across multiple scales, from tree and shrub height to foliage density, greater vegetation heterogeneity provides a greater variety of physical conditions and biological niches and is associated with habitat complexity.

Vegetation structure can be modified by deer in multiple ways (Gill and Beardall, 2001). For example, stem density may be affected by browsing on seedlings, and tree and shrub height and foliage density may be affected by browsing on leading and side shoots. In most cases, deer browsing limits height growth and can delay height growth for several years. Plants that do grow beyond the browsing line may be modified structurally following browsing of side shoots, leading to trees and shrubs with reduced foliage or flowering potential. Usually only the youngest seedlings are killed by browsing, with older trees often withstanding browsing beyond a certain age. In coppice woodlands deer browsing can reduce stem density following cutting and increase light penetration. Although pronounced and prolonged structural changes have the potential to negatively affect many woodland associated species adapted to specific woodland conditions (see Sections 3.4), deer browsing also has the potential to increase species richness or seedling and sapling survival rates by mediating competition from other plants (Gill and Beardall, 2001, Smart et al., 2024). For example, in the recent Bunce survey of Great Britain, under gaps in the canopy created by ash dieback, plant species richness has only increased in areas of high deer grazing, possibly due to the effect of grazing in preventing the dominance of certain forbs species, such as bramble (Smart et al., 2024).

Under low and high deer activity levels (roe and muntjac), vegetation structure was found to be statistically different following coppicing in ancient semi-natural coppice woodlands in central and eastern England (Joys et al., 2004). Following coppice activity, in areas of high deer activity vegetation development was delayed because of browsing pressure, leading to the development of a reduced mid-canopy and extended low shrub layer. For the first four to five years, compartments exposed to high levels of deer activity displayed lower foliage densities (at 0.5m and 1.5m) compared to compartments exposed to low deer activity levels, but this trend was reversed as the mid-canopy developed in areas of low deer activity. A similar effect was observed for bramble, where it was initially more abundant in areas of low deer activity, but eventually became more abundant in areas of high deer activity, possibly because of increased light levels associated with a limited mid-canopy. Additionally, four years after coppicing, the cover of bare ground, litter and moss increased under low levels of deer activity. Taken in the context of natural coppices or coppices undertaken as a conservation intervention, reduced or delayed regrowth has important implications for woodland management.

In the Weald and Welsh Marches high deer (fallow, roe and muntjac) densities (>10 deer km^2) were found to significantly affect the structure of lowland woodlands when compared to low deer densities (~1 deer km^2) (Eichhorn et al., 2017). Woodlands containing high deer densities exhibited 68% lower density of understorey foliage (0.5 to 2m above-ground) than woodlands containing low deer density, while vertical vegetation structure differed consistently between low and high deer density sites, including differences in distribution of foliage and stems in the shrub layer between two metres and five metres, and a greater overall canopy height in high deer density woods. It is possible that the difference in woodland height observed in the high deer density woods is a result of deer browsing itself (Eichhorn et al., 2017). Furthermore, reductions in understorey foliage and woodland structural diversity are likely to impact a range of woodland associated species which rely on specific structural niches (Eichhorn et al., 2017).

Ground flora

Individual species function as components of complexity and can influence ecosystem resilience by increasing species richness and functional redundancy. Excessive herbivory by native and non-native deer can have a profound negative effect on native woodland flora via direct removal of plant material and changes in woodland structure and physical conditions (e.g light regime).

Vascular plants on the woodland floor

Most British woodland communities contain species which are resistant or sensitive to grazing to varying degrees, and which are also palatable to deer to varying degrees, leading to differing outcomes when exposed to high levels of deer grazing. For example, long-term heavy grazing can lead to declines in relatively palatable species such as bilberry and honeysuckle (*Lonicera periclymemum*) and increases in unpalatable species or species resistant to grazing, such as bracken (*Pteridium aquilinum*) or many grass species, respectively (Kirby, 2001). Deer can also browse selectively on flowering material, reducing the reproductive potential of the plant, whereas in plants where flowers are not browsed, reduced vegetative material may reduce flower production (Kirby, 2001), in turn reducing nectar and fruit availability for other taxa. Additionally, damage by trampling can damage certain woodland species such as bluebells (*Hyacinthoides non-scripta*) and dog's-mercury (*Mercurialis perennis*) while bare ground created by deer and the effect of deer on nutrient cycles can favour the colonisation and proliferation of nitrophilous plant species. Changes in light regime, as discussed in the woodland structure section above, can also influence ground flora communities indirectly (Kirby, 2001).

An exclosure experiment in Bradfield Woods in Southern England, an ancient semi-natural woodland with a long history of coppicing, found species richness and diversity were significantly greater in grazed controls than in ungrazed exclosures (Stone et al., 2004). This effect can be partly explained by a reduction in bramble cover outside of exclosures, which was negatively correlated with species richness. Similar results have been reported in the recent Bunce survey of Britain and Ireland (Smart et al., 2024) where deer mediated reductions in bramble cover in ash dieback canopy gaps have led to increased species richness. Roe, muntjac and fallow deer were present in Bradfield Woods at the time of the study. Although a greater overall species richness and diversity was observed in grazed control plots, the long-term effect on flowering and seed production of plants in the controls, as well as the impact on floral species of interest, is not known.

Additional studies add further context to the relationship between deer herbivory and species richness. Following historical declines in bramble and woodland forbs and increases in grass species, another exclosure experiment in Wytham Woods in Oxfordshire found that forb species began to recover inside exclosures compared to the wider woodland containing roe, fallow and muntjac deer (Morecroft et al., 2001). In the same study, exclosures created in woodland gaps were rapidly colonised by bramble whereas bramble cover remained low in exclosures under the canopy. In another exclosure study in Dorset looking at the effect of roe deer on vegetation density in small farm woodlands, density of vegetation cover was reduced by deer browsing in summer and winter. Furthermore, the vegetation composition had changed significantly by the end of the four year study period, with unfenced plots containing more grass and bare ground and less cleavers than fenced exclosures (Sage et al., 2004).

Non-vascular plants on the woodland floor

There are fewer studies investigating the effect of deer on non-vascular plants and lichens, which also form key components of woodland communities, although there is evidence that grazing by red deer on boulder tops in Scotland can prevent ecological succession to the benefit of saxicolous bryophyte and lichen communities (Moore and Crawley, 2014).

There is further evidence from North America to suggest that deer mediated vegetation changes may benefit bryophytes to some extent. In Haida Gwaii, an island archipelago off the coast of British Columbia dominated by temperate conifer woodland, introduced black-tailed deer (*Odocoileus hemionus*) have drastically reduced tree regeneration and extent of understorey vascular plant cover since their arrival. In a study investigating bryophyte response to deer herbivory, bryophyte cover was higher on islands with deer than on islands without deer, suggesting that deer mediated vegetation changes are benefiting bryophytes, either through reduced competition for resources or via a changing light regime. Furthermore, bryophytes were avoided by black-tailed deer even under severe browsing pressure (Chollet et al., 2013). Haida Gwaii has a comparable climate to much of the UK, and it is likely that these findings are applicable to British woodlands.

Fauna

Birds

Deer can directly affect woodland structure and ground flora communities and have knock-on effects on other woodland associated taxa. As with the ground flora species outlined above, these species perform ecological functions from which resilience can emerge. The effect of deer on woodland birds is well studied in woodlands in the UK and several studies indicate that high levels of deer browsing can alter habitat structure in such a way that negatively affects several bird species, primarily those associated with the woodland understorey (Gill and Fuller, 2007). The removal of low woody vegetation by deer can result in loss of nesting sites, exposure to predation and reduction in food resources for several species (Gill and Fuller, 2007).

In lowland England, Newson et al. (2001) provide evidence for a negative association between roe, fallow and muntjac deer and several woodland bird populations associated with dense understorey habitats, using extensive national bird and deer monitoring data (Newson et al., 2001). This study only provides evidence of a correlation between increasing deer populations and reductions in many woodland bird species though, and the results should be considered in conjunction with other proposed drivers of declines in woodland birds.

Several studies have been undertaken investigating the effect of deer browsing on woodland birds in Bradfield Woods in Eastern England, a relatively small woodland surrounded by farmland, and supporting populations of roe, muntjac and fallow deer. Deer exclusion has been shown to benefit male blackcap (*Sylvia atricapilla*) body condition, singing incidence and timing of singing (with earlier singing observed in non-browsed plots) (Holt et al., 2013), with potential implications for population productivity. Similarly, greater nightingale (*Luscinia megarhynchos*) territory density was recorded inside exclosures in a separate experiment (Holt et al., 2010), suggesting that deer populations can negatively affect breeding nightingales.

Another exclosure experiment in Bradfield provides evidence that both vegetation age and deer browsing affect bird assemblage composition and structure in both spring and winter (Holt et al., 2014), with shrub-layer foraging species recorded less frequently in older and browsed vegetation in both spring and winter. Ground foraging, ground nesting and shrub-layer nesting species were also recorded less frequently in browsed vegetation, although no effect of vegetation age was reported (Holt et al., 2014). Certain species, such as robin (*Erithacus rubecula*), and wren (*Troglodytes troglodytes*), were recorded less frequently in browsed areas only in winter. This is important as past studies may have underestimated the effect of deer browsing on woodland birds if only studying the effect on breeding birds. In a mist-netting experiment at Bradfield Woods using the same exclosures, more ground and understorey foraging birds were captured where deer were excluded (Holt et al., 2011), while no significant positive responses to browsing were detected. No effect of deer browsing on habitat use by woodcock in winter was observed in another study at Bradfield Woods (Holt and Fuller, 2013).

Mammals

Deer can impact small mammal populations in two main ways. Namely, the modification or removal of habitat and the increased competition for resources following deer herbivory (Flowerdew and Ellwood, 2001). Changes in different layers of woodland vegetation can affect different species in different ways. For example, bank voles (*Myodes glareolus*), which favour dense ground vegetation within woodlands, may benefit from a lack of tree regeneration and extended open periods in canopy gaps grazed by deer. By contrast, if deer browsing reduces the occurrence of understorey species such as hazel (*Corylus avellana*), dormouse (*Muscardinus avellanarius*) and yellow-necked mouse (*Apodemus flavicollis*), which favour mature deciduous woodlands and shrubby understories containing hazel, may experience population reductions. Furthermore, field voles (*Microtus agrestis*) and shrews (*Sorex* spp.) may benefit from deer grazing in grass dominated woodland rides and glades (Flowerdew and Ellwood, 2001).

In Wytham Woods, historical bank vole population declines are thought to be linked to deer grazing, as wood mouse (*Apodemus sylvaticus*) populations, which do not require the same level of dense understorey vegetation, have not declined in the same way in the face of structural vegetation changes resulting in a reduced understorey and ground layer (Macdonald, 2015). In an exclosure experiment in Wytham Woods in the early 2000s bank voles were found to be more abundant inside exclosures compared to adjacent deer-grazed transects containing significantly reduced shrub layers (Buesching et al., 2011). Wood mice were more frequent outside of exclosures, however overall rodent abundance was greater inside deer exclosures, suggesting a higher overall rodent carrying capacity in areas exposed to lower deer pressure. A follow-up experiment in 2010 provides encouraging signs that small mammal populations can recover relatively quickly following a reduction in deer pressure. Management at the site successfully reduced deer density from between 0.4–1.6 deer per hectare in late 1990s to 0.17 deer per hectare from 2003 to the time of the study (Bush et al., 2012). Following management vegetation recovered outside of fenced exclosures (albeit at a slower rate due to maintained levels of low grazing) such that in 2010 bank vole and wood mouse numbers showed no difference between treatments and appear to have recovered from the effects of heavy grazing.

Invertebrates

There are relatively few studies looking at effect of deer on invertebrates in Britain. In Scotland, however, invertebrate abundance has been found to be higher in deer exclosures than in unfenced areas grazed by red deer (Baines et al., 1994). Across eight native pinewoods in the Scottish Highlands, geometrid moth larvae were negatively associated with red deer grazing, possibly due to a competitive effect whereby deer graze the nutritious tips of bilberry with which they are associated. As geometrid moths are most abundant on bilberry, which is associated with intermediate levels of tree mediated shading, mature forest stands of 200 to 300 trees ha^{-1} grazed by <5 deer km^{-2} are recommended to maximise bilberry cover and moth abundance.

In a review paper by Feber et al. (2001), various examples of the impact deer may have on butterfly species are provided. While herbivory is noted to benefit a range of butterfly species via a modification of habitat structure, reported negative effects include the reduction in suitable egg-laying material for white admiral (*Ladoga Camilla*) following browsing of honeysuckle by muntjac and a reduction in suitable growth periods following coppicing for several early successional species, due to the requirement to fence off coppices after cutting. By contrast, recently cut coppice left unfenced can lead to high levels of damage to stools and the development of swards dominated by grasses and sedges, which will also reduce habitat quality for several species.

Positive effects of grazing include the maintenance of open rides and glades required by many species. In particular, pearl-bordered fritillary (*Boloria euphrosyne*) in western Scotland has been shown to benefit from a lack of tree regeneration maintained by grazing. While there is evidence that exclosure can benefit pearl-bordered fritillary in the short term due to the development of a taller sward containing a greater abundance of preferred nectar sources, in the longer term some grazing is likely to be required to maintain the open conditions preferred by the species. Additionally, roe deer are considered an important species in Scotland for maintaining areas for chequered skipper (*Carterocephalus Palaemon*) (Feber et al., 2001).

Nutrient cycles and seed dispersal

Browsing by large herbivores, including deer, can have significant effects on ecosystem nutrient cycles, which has important implications for the trajectory upon which a vegetation community is able to develop. In an exclosure study in Creag Meagaidh national nature reserve in the Scottish Highlands, browsing at relatively low density by red deer (<5 deer km-2) was found to have significant negative effects on soil biological properties and nutrient dynamics. In exclosures where deer were not present, a positive feedback loop was observed whereby increased tree growth (of the nitrogen fixing *Betula pubescens*) stimulated carbon and nitrogen mineralisation, which further increased plant nitrogen supply and tree growth (Harrison and Bardgett, 2004). This finding has important implications for landscape scale natural regeneration and woodland productivity in the uplands, although the effect significantly higher deer populations than those reported in this study may have is currently unknown. As the results reported here relate to what is generally considered a near-sustainable population of a native species, it is possible that at a certain level negative effects on soil biological properties and nutrient dynamics should be accepted as an acceptable trade-off of having deer in the landscape, which is clearly beneficial in sustainable densities for other components of complexity.

A subsequent study utilising the same exclosures found that removal of deer changed the relative availability of nitrogen and phosphorous and led to a shift from nitrogen to phosphorous limitation. This is likely due to increased nitrogen mineralisation, but not phosporous, in deer exlosure plots (Carline et al., 2005). Furthermore, litter of un-browsed trees decomposed faster than litter from browsed areas in the same study area (Harrison and Bardgett, 2003), regardless of soil origin, suggesting that browsing by deer, rather than the effect of deer on soil biological proprteries, is the key factor influencing enhanced decomposition of birch litter in this ecosystem. These results suggest that the negative effects of deer on soil biological properties and nutrient cycling are at least in part related to changes in litter quality (Harrision and Bardgett, 2003). However, it is important to note that these studies cover a single ecosystem and results may differ in different woodland types across the UK.

Kirby (2001) provides a brief assessment of some of the more localised effects of deer on soil nutrient status due to the consumption of plant matter in one area and dunging on others (although specifics regarding population densities in relation to the magnitude of effects are not included). Over time preferential browsing on nutirent rich vegetation may lead to an increase in nitrophilous plant species. Indeed, the release of dung and urine returns labile nutrients to the soil, but may be countered by a reduction in soil biological acitivty following grazing by deer, as discussed above (Harrision and Bardgett, 2004).

Furthermore, while it is recognised that large herbivores play an important role in the dispersal of ingested seeds, few studies have directly investigated the effect of deer-faciliated seed dispersal in woodlands in the UK. However, Eycott et al. provide evidence that deer may play an important role in dispering plant seeds in a woodland mosaic in Thetford forest in eastern England (Eycott et al., 2007). By examining the contents of faecal pellets for a range of deer species (red, roe, fallow and muntjac), they found that deer dispersed at least 101 plant species, a third of which lacked any other dispersal mechanism. Importantly, although the total seed input from deer is low in relation to the total seed rain, the role in deer in dispersing seeds which would not otherwise be dispersed may play an important role in community structure, for example by moving seeds from plants growing under a closing canopy to an area of early succesional woodland. Deer species also differed significantly in their dispersal activity, with large-bodied grazing species (red and fallow deer) dispersing a greater number of plant species than smaller-bodied browsing deer. Despite its numerical dominance in the area, muntjac deposited the fewest seeds per gram of any deer species. There is no available data on dispersal of plant meterial in deer fur, or to what extent deer play a role in the dispersal of invasive or ruderal species, although this is also likely to occur. These mechanisms again provide evidence that deer play a vital role in woodland functioning, and that sustainable numbers of deer are vital to maintaining woodland resilience.

Climate

The potential threats to woodland resilience associated with the effect of unsustainable deer populations on the ecological factors outlined above are likely to be exacerbated by the effects of climate change. However, despite the known threat of a rapidly changing climate, there is relatively little available literature explicitly exploring how deer populations may interact with climate change.

A study at Mar Lodge estate in the Cairngorms, Scotland, found that red deer herbivory and climate had significant interactive effects on tree growth (Vuorinen et al., 2020). Scots pine growth responded more strongly to annual temperature in the presence of red deer, and although growth was negatively correlated with deer density and positively correlated with temperature, warming decreased tree growth when more than 60% of shoots were browsed, again highlighting the importance of maintaining sustainable deer numbers when aiming to facilitate increased natural woodland regeneration. At lower deer densities, pine individuals growing in the presence of deer may be able to tolerate browsing better in a warming climate due to an enhanced growth response to temperature, possibly via reduced plant-plant competition (Vuorinen et al., 2020).

Additionally, a recent evidence review investigating the potential effect of deer on Scotland's carbon sequestration goals found that while potential mechanisms through which deer may affect carbon cycling have mostly been identified, there is limited evidence available which examines the magnitude of these effects (Hirst, 2021). Proposed mechanisms through which deer may affect carbon cycling include both direct and indirect effects. Direct effects include consumption of plant biomass that would otherwise photosynthesise and store carbon, browsing which reduces natural regeneration, death of larger trees due to damage caused by fraying and trampling and soil compaction, which has the potential to affect carbon cycling. Indirect effects include a promotion of the dominance of unpalatable plant species and a subsequent effect on decomposer communities, and a potentially beneficial effect on decomposer communities as a result of defecation and urination. Although not currently quantified, these effects, taken alone or in combination, are likely to pose significant threats to carbon sequestration goals.

High deer densities can also alter the ground vegetation on peatlands and in severe cases promote erosion by creating bare patches of exposed peat (Cummins et al., 2011). In the context of landscape-scale conservation efforts, where woodlands are managed in dynamic mosaics with other habitats, deer-mediated peat erosion has important implications not only for the UK's resilience to climate change, but for the resilience of specific vegetation communities with low-density tree cover which require undisturbed peat.

Management

Identifying sustainable deer numbers

Artificially high deer densities in the UK negatively impact several ecological functions likely to be linked with long-term woodland resilience. However, it is also clear that at more sustainable population levels, deer play a key role in maintaining and enhancing woodland dynamism by enabling important ecological functions. It is therefore imperative that sustainable deer populations are identified to maintain or restore healthy woodland function. Importantly, deer damage cannot be reliably predicted as a function of deer density alone. The density of deer required to make damage to woodland likely differs between species (Putman et al., 2011) and is also influenced by interactions between climate and landscape factors such as woodland cover (Spake et al., 2020, Reid et al., 2021), and the overall management approach for the site. For instance, conservation grazing with large domestic herbivores can be an important part of land management, but where this is considered alongside high numbers of browsing deer, care is required to ensure impacts don't become detrimental to objectives.

As such, sustainable populations should be defined on a site-by-site basis (with the site being set in the context of a larger landscape-scale partnership, see below) and in line with local conservation objectives. In recent decades, difficulties and costs associated with estimating true deer densities meant it may often have been desirable to base these sustainable populations on assessment of impacts of deer, alongside estimates of actual density (Putman et al., 2011). However, rapid advances in drone technology are now making estimates of actual density possible at an unprecedented level of accuracy.

Management at scale

Understanding scale

The management of sustainable levels of deer populations across the UK should be at the forefront of woodland protection, restoration and expansion targets in the coming decades. Indeed, in areas of particularly high deer densities, funding directed towards deer control will be as important as funding for woodland creation sites, which will likely be subject to significant damage without mitigation, and encouraging uptake of such funding by landowners will be as important as the provision of the funding itself. Ineffective deer control at new woodland sites may also raise concerns over use of public money (for woodland creation) and jeopardise woodland creation goals in the long term.

In most of the UK, deer numbers are controlled by stalking, usually on a part-time or recreational basis. But to be fully effective, such management should be coordinated at landscape scale (Putman, 2011). In practice much needs to be done to translate the current understanding of deer ecology and deer impacts into successful management practices. The iDeer Project (funded by the Future of UK Treescapes Programme and led by the University of Reading), aims to bridge this gap by developing an interactive decision support tool which will help to minimise deer impacts on new and existing woodlands in England and Wales. Crucially, the production of 'risk maps' will allow neighbouring landowners to understand how a management intervention on one land parcel, for example the erection of deer fence, may influence deer impacts on another land parcel. The project will allow for the development of management plans informed by both local and scientific knowledge, and reduce the risk of landowner conflict, allowing for more integrated and cooperative management at the landscape scale.

The concept of 'landscape scale' is subjective which can lead to debate regarding at what scale cooperative management is required. Recently, data used to quantify reductions in deer damage in response to culling at multiple scales has become available, providing evidence that increasing culling area can reduce the impacts of multiple deer species (Fattorini et al., 2020). Using woodland impact assessment and deer harvest data from 98 woodlands, modelling indicated that for roe and muntjac, which are both relatively smaller bodied and territorial species, culling at the local level (within 2.5km of target woodlands) was effective in reducing impact, although small increases in effectiveness were expected by increasing the culling radius to between 30km and 70km. For red deer and fallow deer, both larger bodied and herding species with larger ranges, culling was only effective at above the single woodland scale, although modest effects were observed by culling red deer at the local level. The effects of coordinated management extended to a radius of 100km, although in practice, such large areas are unlikely to be feasible. The study took place in a lowland and relatively well-connected woodland landscape in East Anglia, where deer occur at moderate to high density. In landscapes where dispersal of deer is more difficult, due to geographic or landscape factors, the ranges required to adequately control deer numbers may be reduced.

Several landscape-scale efforts in Scotland indicate that sustained culling has successfully enabled woodland regeneration and expansion in much smaller areas than the maximum reported management area in Fattorini et al., (2020). For example: the 29,000ha Mar Lodge estate (Rao et al., 2017), the 23,000ha Corrour estate (Watts, 2024) and the 18,000ha Glen Feshie estate (Gullet et al., 2023). Further work to identify the minimum scale at which coordinated deer management is likely to be successful across different landscapes will be beneficial.

Case studies

High Weald Deer Management Project

In practice, several examples now exist in which landscape-scale cooperation is being utilised. The recently completed High Weald Deer Management Project for example, led by the High Weald National Landscape Partnership with funding and support from the Woodland Trust and DEFRA, was set up with the aim to bring landowners and deer stalkers together to form local deer management groups in two target areas covering 230km^2 (Williams, 2024). Drone surveys funded by DEFRA's Farming in Protected Landscapes grant have allowed for estimations of the local deer population, which have revealed an average deer density of 20 to 25 deer per km^2, and up to 50 deer per km^2 in some areas. While payment was claimed for more fallow does in 2024 than in 2023, (20 landowners claimed for 324 does in 2023 compared to 30 landowners claiming for 617 does in 2024), these numbers appear unable to currently meet the 30% (8,766 deer) reduction in fallow deer numbers required to maintain a static population, or 40% (12,000) required to bring the population down to the desired density of <10 deer per km^2, highlighting the scale of the issue. The significant rise in landowners claiming and total number of deer culled is encouraging, however a reduction in deer numbers to the desired density will only be achievable with sustained and increased culling over at least 10 years. Additional partnerships exist which aim to highlight the importance of managing deer at a landscape level, such as the Deer Initiative, which aims to achieve and maintain a sustainable and healthy population of wild deer in England and Wales. Ultimately, the success of this endeavour will rely on the upscaling and prioritising of landscape-scale deer management (The Deer Initiative, 2024a).

Cairngorms Connect

Further evidence of large-scale concerted culling efforts having a tangible effect on deer numbers and tree regeneration can be found in the Cairngorms in the Eastern Highlands (Gullet et al., 2023). The Cairngorms Connect landscape restoration partnership covers 60,000ha and is ran cooperatively by four landowners comprising Forestry and Land Scotland, NatureScot, the RSPB and Wildland Ltd. While fences have commonly been used to exclude deer from regenerating woodlands, they can limit the movement of other native species and are costly to erect. Additionally, the complete removal of deer is unlikely to provide optimal conditions for the development of resilient woodlands. The Cairngorms Connect project instead utilises deer culling at landscape scale without the use of fences. During peak periods of regeneration, this has resulted in an increase of approximately 164ha of new woodland per year, providing clear evidence that collaborative deer management across multiple landholdings can achieve rapid landscape-scale woodland expansion without the need for planting or fencing. The project provides evidence that woodland regeneration can occur at red deer densities below $4km^2$ in native pine woodlands, however there is evidence that the most palatable tree species may require periods of lower deer densities to successfully regenerate. The work also highlights the potential for adaptive management to inform the level of cull required to allow for woodland regeneration, and the possibility of a mixture of intense pulse culls and periods of relatively low-level culling to provide the conditions necessary for long-term sustainable woodland management. Further evidence that sustained deer culling at the landscape scale can significantly reduce deer numbers and increase woodland regeneration rates comes from the Mar Lodge estate, also in the Cairngorms, where similar results have been reported (Rao, 2017).

Understanding demographics

It is widely accepted that targeting female deer is desirable in order to limit reproductive capacity of a given deer population, as advocated by the Deer Initiative's best practice deer culling guidance (Deer Initiative, 2024b). The guidance states that an initial reduction cull aimed at reducing unsustainable deer numbers should focus on achieving the target cull of female deer, which in cases where deer are causing unacceptable habitat damage, can be set at the largest number of deer that can be humanely killed each year. Males may also be taken outside of the female season. As stalkers selling venison are paid according to the weight of the carcass they provide, there may be a temptation to shoot males (which are heavier) inside the female season. However, this should be avoided if maximising population control efficacy is the aim. As such, developing incentives which enable stalkers to prioritise female deer is key to maximising the efficacy of deer population control through stalking.

Following an initial reduction cull, maintenance culls are required in order to maintain a more sustainable population. Here, the cull for each age/sex class should be similar from year to year. Assuming that sex ratios are approximately equal, approximate recommended cull rates can be generalised as follows: red, fallow and sika populations will require a cull of at least 20% of the population, while roe, muntjac and Chinese water deer will require a cull of around 30% of the population, with at least half of the cull for each species comprising females. Minimum and maximum cull figures should be set for breeding females and other sex and age classes. If the minimum breeding female cull figures are achieved, the numbers and ages of males culled can be flexible to suit objectives.

In addition to the information provided in the Deer Initiative's best practice deer culling guidance, Forest Research has also published an interactive deer model which allows deer managers to predict the effect of culling on future deer populations in order to set appropriate cull targets (Forest Research, 2022), although this was not accessed during the review.

Next steps

What needs to happen

It is essential for our woodlands that sustainable deer population thresholds are identified for different contexts, and management is able to achieve and maintain such thresholds. New planting, restocking and woodland creation will be a challenge without significant investment in time and money to protect that stock, especially as the sector moves away from the use of plastic tree shelters.

Areas to focus on include:

- Better training and support for deer managers and landowners.
- Recruitment of new deer stalkers to improve capacity and diversify the sector.
- A simplified licensing system to support access to night licensing in difficult to control areas.
- Incentives and government funding to enable landscape level control and

targeted culling demographics.

- Continued development of the sustainable use of venison as a high value meat product and avoidance of markets being bottlenecked by a lack of demand for carcasses.

While an uplift in scale and effort is required across the UK to reduce deer numbers to sustainable levels, landscape-scale projects such as the High Weald Deer Management project or the Cairngorm Connect project provide evidence of what is possible. A key issue in the coming decades will be the upscaling and connecting of such efforts. The results of these projects can be used to unlock funding and inspire the creation and uptake of such partnerships across the UK.

In the meantime, furthering our understanding of deer ecology and effective management through continued research is essential. In the longer term, manipulation of the factors responsible for deer overabundance are likely to be key factors in returning deer populations to sustainable levels. An ambitious option here is the reintroduction of large predators. While there remains conflict between stakeholders and large predators in mainland Europe, the increase in numbers of various large predators on the continent as a result of natural expansion and conservation reintroductions can be seen broadly as a conservation success story. As an archipelago, the UK will be without a potential agent of population control until reintroduction measures are seriously considered. Although the UK will pose its own specific challenges to reintroduction, such as the density of the population present and the public perception of apex predators following centuries of their absence, the reintroduction of such predators to the UK should be taken seriously as a potential management tool for deer populations.

Recently, van Beeck Calkoen et al. (2024) found that in human-dominated systems in Europe, human hunting plays a greater role than presence of predators in reducing red deer density and that density was more strongly limited by predators at sites with lower level human land-use (van Beeck Calkoen et al., 2024). The study also reports that red deer density was not significantly reduced where only wolf (***Canis lupus***) and lynx (***Lynx lynx***), and not brown bear (***Ursus arctos***), were present. As predator populations expand throughout Europe it is possible that this dynamic may change over time. Furthermore, potential benefits of predator reintroduction extend beyond direct predation, through the creation of a landscape of fear, whereby deer actively avoid areas or browse less where there is a high risk of predation (Manning et al., 2009). However, there has been some criticism of this concept as beyond small-scale behavioural effects (e.g. changes in activity patterns throughout the day) evidence is scarce and very much inconclusive as to sustained impact. Nevertheless, the introduction of natural predators of deer has profound implications for woodland landscapes. While the reintroduction of the wolf is not on the immediate horizon in the UK, introductions of smaller predators capable of taking deer, such as the lynx, may be more feasible, although careful consideration is needed to ensure that sufficient contiguous woodland habitat is available to support viable lynx populations and that suitable processes are in place to minimise human-wildlife conflict (Milner, 2015).

Together, the expansion of landscape-scale deer management projects, continued research into deer ecologies and impacts, and eventual restoration of natural processes such as predation, provide a pathway to sustainable deer numbers and ultimately, resilient woodland landscapes. Indeed, concerns surrounding the dwindling capacity and ageing demographic of the deer stalking community, in combination with the currently limited options for the end use of deer carcasses, could mean that predation, and enthusiasm for its role in deer management, comes to play an increasingly important role in deer management in the coming decades.

Recommendations for future research

The following evidence requirements and gaps were identified while conducting this review:

- Enhanced accuracy of deer population estimates across Scotland, England, Wales and Northern Ireland. With new technology, such as drones and artificial intelligence improving survey efficacy and efficiency, a more accurate estimation of the total and species-specific deer populations in the UK will help inform policy and management.
- Further investigation into deer population density thresholds above which natural regeneration and colonisation can occur, for different species of deer in different woodland contexts, is needed. However, evidence-based decisions regarding suitable thresholds should be made on both a site-by-site and landscape-scale basis.
- The spatial scale at which deer management results in sustainable and successful outcomes for woodland creation and regeneration, and over what time periods, in different woodland contexts, is still largely unknown. Part of this is an increased understanding of how deer move in and out of woodland areas and landscapes at different temporal scales (e.g. days, weeks, seasons).
- The impacts of different species of deer, and the extent to which native and non-natives differ, on trees and woods in different contexts is required to enable better targeting of management.
- Further research quantifying the effect of deer impacts at different densities on carbon sequestration and other ecosystem services is needed.
- Best practice guidance for deer management is available. However, the nuance of how deer management interacts with other woodland management interventions and decisions is largely unknown. For instance, interactions with conservation grazing objectives.
- Existing deer data collection across the sector would benefit from being reviewed, and a coordinated, strategic approach to data collection adopted into the future. The Woodland Trust data presented here is just one example, and joined up data collection and analysis is required to allow effective decision making and analysis of impact at the landscape, regional and national scale to inform future management and policy decisions.
- The extent to which reintroductions of lost predators are feasible, acceptable to communities and necessary to enable sustainable cessation of deer as a threat to UK woodland resilience.

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Pollution

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Abstract

Pollution is an underappreciated threat to woodland integrity and a driver of systemic change, acting on every level from soil chemistry to species dynamics. The fragmented nature and large edge area of UK woodlands heightens their susceptibility to environmental pollution. The impacts of pollution are often cryptic, and often subject to biological and chemical lags. Despite gradual reductions in exceedance, critical loads for excess acid and nutrient nitrogen (N) are still widely exceeded across UK woodland habitats, with hotspot regions in agricultural intensive regions such as Northern Ireland, the Welsh Borders and East Anglia. Owing to reductions in sulphur pollution and the continued expansion of intensive agriculture, concentrations of ammonia (NH_3) are increasing in every region of the UK, including for protected sites such as SSSIs. Monitoring of forest sites across the UK and Europe suggests that conifer species have lower physiological tolerance to excess N and have a generally increased likelihood of crown damage. To date, excess N has been broadly 'positive' for UK broadleaves in terms of growth, and major shifts in condition have not been observed. However, emerging evidence suggests that chronic exposure to excess N can be a stressor for both conifer and broadleaf tree species, and N deposition is positively related to the prevalence of acute oak decline (AOD). Excess N also modulates responses to environmental factors such as drought. Long-term monitoring of woodland sites across the UK indicates woodlands have responded to excess N with shifts towards plant species of higher fertility, and changes to mycorrhiza and other soil microbiota are also indicated. Lichens and bryophytes are particularly sensitive to NH3, and ongoing impacts are expected across the UK where the average concentration exceeds 1 µg m-3. Impacts of toxic N compounds (e.g. ammonium, NH4+) on sensitive invertebrates are also likely but under-investigated. Evidence of tree nutritional deficiency has now been identified in Europe and in the UK, hypothesised to be caused by excess growth or changes to mycorrhiza. Existing N critical load thresholds for broadleaf woodlands (10-20kg N ha^{-1} yr^{-1}) are likely insufficient to protect the most sensitive species and processes. Continual exceedance of critical N loads risks N saturation for UK woodlands in heavily polluted regions; this eventually may result in reaching an ecological tipping point and irreversible changes to condition, form and function. Tropospheric (ground level) ozone (O_3) also affects UK woodlands. On an annual basis, concentrations of O_3 exceed a critical level of 5mmol m^{-2} in many regions of the UK, causing an estimated mean 7% biomass reduction for managed broadleaf habitat in 2019. In the urban realm, levels of particulate matter (PM) are slowly decreasing but are a significant threat to human health and are also a stressor for vegetation; urban trees can help control air pollution by blocking polluted air masses and can also remove a small amount of pollution, though this latter effect is minor on a local or city level. The effect of chronic PM exposure on the health of urban woods and trees is virtually unknown but could be a significant pressure near major pollution sources such as busy roads and airports. In addition

to the above, UK woodlands are also exposed to a range of other pollutants including heavy metals, persistent organic pollutants (POPs), pesticides and microplastics. Fly-tipping is also a growing problem across the Woodland Trust estate. In all cases, a lack of data and monitoring hinders more complete understanding of their current extent and impacts. Levels of pollution will remain above critical load/critical level thresholds for the foreseeable future and will play a significant role in moderating the response of woodlands to the effects of climate change.

Highlights

- Pollution impacts woodlands at every level – including soil chemistry, plant biochemistry and physiology, woodland structure and species dynamics.
- Total N pollution may have peaked due to improved control of emissions; however, exposure to excess acidity and nutrient N still widely exceeds critical loads for UK woodlands. The level of exceedance for woodland habitats is slowly decreasing in every nation apart from Northern Ireland. Reactive N compounds are now the major cause of woodland acidification.
- Average concentrations and exposures of ammonia are increasing in every region of the UK, this includes for a range of woodland habitats and protected sites. This is due to reduced scavenging of reduced N by sulphur, but also the continuing expansion of point source pollution such as intensive poultry units (IPUs).
- Conifers are generally more sensitive to excess N. However, continual excess N can enhance sensitivity of some broadleaf species to pest, disease or environmental stress; N deposition is positively associated with acute oak decline (AOD). N interacts with temperature and drought effects and will play a major role in moderating the responses of woodlands to climate change.
- The Bunce survey, monitoring 50 years of change across UK broadleaf woodlands, has found a shift in vegetation towards species of higher fertility; the long-term effects of N pollution may be masked by changes such as increased canopy densities and lower light levels. Changes to the composition of mycorrhiza and soil microbiota are also indicated.
- Loss of sensitive lichen and bryophyte species is expected to be widespread and ongoing due to toxic ammonia, but monitoring is lacking; lichens may be at an increasing competitive disadvantage in some woodlands.
- Emerging evidence suggests that some forests across Europe and UK may be beginning to experience nutritional deficiencies, perhaps due to excess growth/changes to mycorrhiza. Continuing exceedance risks N saturation, and irreversible changes to condition and function.
- Some changes to soil chemical profiles/soil microbiota are likely to be permanent; the effects of enrichment due to Roman activity are still apparent after 2,000 years.
- Current critical loads for N (10-20kg ha^{-1} yr^{-1}) are likely insufficient to protect most sensitive species and processes – particularly for sensitive conifer woodlands.
- Ground level ozone (O_3) is a phytotoxin and poses a threat to the health of

woodland species; responses vary on a species-specific level; it is predicted to impact biomass accumulation in woodlands on an annual basis in some UK regions; sensory or respiratory impacts on insects and even birds have also been documented in literature and cannot be ruled out. O_3 also moderates the response of plants to drought and other factors.

- Particulate matter comprises a variety of compounds and is the most important air pollutant for human health, causing thousands of premature deaths in the UK on an annual basis. It is also a known stressor for trees and vegetation; however, the effect of chronic exposure for the urban forest is virtually unknown.
- Woodlands are affected by a range of other pollutants including heavy metals, pesticides, microplastics and persistent organic pollutants; in all cases, a lack of monitoring hinders more complete understanding of their current extent and impacts.
- Fly-tipping is a growing problem on the Woodland Trust estate due partially to increasing poverty/decreasing availability of recycling and disposal facilities.
- Suggested policy improvements include (amongst others) improved awareness of policy makers around air pollution issues, tighter regulation of IPU permissions, improved control of O_3 precursors and reversal of post-Brexit changes to pesticide policy.
- Key evidence gaps (amongst others) include species-specific and ecosystem-wide impacts of pollutants, research into impacts of emerging pollutants (e.g. microplastics) and costs associated with damage to woodland ecosystem services.

Introduction

Historically, the UK has experienced profound environmental harm due to pollution, including to woodland habitats. Since the 1970s, there have been substantial reductions in the emission of a range of major pollutants on a UK-wide basis, including total nitrogen, some heavy metals, persistent organic pollutants and sulphur (RoTAP, 2012; Guerreiro et al., 2014; APIS, 2016b; Rowe et al., 2023; DEFRA, 2024b; ONS, 2024). Despite declining trends for many pollutants, atmospheric pollution remains a major driver of ecosystem change and poses a significant threat to our native woods and trees. Although pollution is intangibly related to other human impacts – and often on a global basis - it also presents a simultaneous set of local stress factors on our natural and managed ecosystems. In the following chapter, an updated review of major pollutants assessed under the auspices of the UN Long-Range Transboundary Air Pollution Convention (CLRTAP) European Monitoring and Evaluation Programme (EMEP) (CLRTAP, 2017), as well as national legislation such as the UK Government Clean Air Act (DEFRA, 2019), is provided: their trends and status in the UK and a summary of their current impacts on UK woodland ecosystems where this information is available. Future impacts of major pollutants, in the context of global change, are considered. The role of other pollutants, such as pesticides and microplastics, is also highlighted. Water pollution, recently assessed in the River Trust's State of Rivers

Report (The Rivers Trust, 2024), is not considered here but is known to be a significant factor affecting the health of wet woodland habitats (Wildfowl and Wetlands Trust, 2024).

Methods

A non-exhaustive literature review was conducted to provide updated information on major pollutants monitoring in the UK and internationally. Searches primarily focused on UK studies but also included those from other regions or from a global perspective where necessary (for example, global transport of ozone precursors). No date restrictions were used, but more recent research (post 2010) was prioritised. Searches for primary evidence were conducted using Google Scholar and included primary research as well as grey literature and websites of key resources or datasets. In combination with standard Boolean operators, Keyword search terms included amongst others: 'woodland' 'critical loads' 'critical thresholds' 'nitrogen deposition' 'ammonia' 'ozone' 'particulate matter' 'microplastics', 'pesticides' 'UK air pollution' 'Impacts' 'pH' 'acidification' 'broadleaf' 'conifer' 'heavy metals' 'CLRTAP' 'vegetation' 'lichens' 'bryophytes' 'health impacts' 'birds' 'soil fertility' 'pests' 'disease' and 'stress'.

Results

Nitrogen and acidity

Sources and current trends

Environmental nitrogen (N) pollution is produced from heavy industry, transport and intensive agriculture. N deposition impacts natural habitats through eutrophication (enrichment), changes to pH as well as directly toxic effects of N compounds. Since 1990, there has been a long-term decrease in overall UK N pollution (Worrall et al., 2016; Tomlinson et al., 2021). This trend has been primarily affected by reducing inputs of synthetic fertiliser and improving the efficiency of industrial and combustion processes, leading to reduced losses to rivers and the atmosphere (Worrall et al., 2016). On a national scale, oxidised forms of N (nitrous oxides, NOx) remain the single largest source of atmospheric N pollution despite substantial reductions (Hicks et al., 2022) (Table 1). Emissions of reduced N such as ammonia (NH_3) have remained relatively stable and are currently increasing, reflecting their primary source in agriculture and food production as well as decreased chemical scavenging by sulphur compounds (Warner et al., 2017; Hicks et al., 2022). Within the agricultural sector, the largest sources of NH_3 are from direct soil emissions, cattle and wastes (Hicks et al., 2022). Agricultural soil is also the largest source of nitrous oxide (N_2O), a potent greenhouse gas (Hicks et al., 2022). Emissions of sulphur (S), historically an important contributor of acid rain, are substantially reduced on a UK-wide basis, and N pollution is now the primary cause of ongoing acidification or alkalinity across woodland habitats (Rowe et al., 2023).

Table 1: Major forms of nitrogen pollution

Type of deposition	Reduced	Oxidised
Wet and occult deposition (aerosol, clouds, fog, rain or snow)	Ammonium (NH_4^+)	Nitrate (NO_3^-)
Dry deposition (gas)	Ammonia (NH_3)	Nitric acid (NHO_3); Nitrous oxides (NO_x)

Since the previous *State of the UK's Woods and Trees* report (Reid et al., 2021), a review and recalculation of critical loads using updated evidence and changes to habitat mapping have led to a substantial increase in modelled exceedance statistics in the most recent air pollution trends analysis performed by UKCEH (Rowe et al., 2020; Bobbick et al., 2022; Rowe et al., 2023). In the 2019-2021 period, deposition of acidity and nutrient N (derived from the sum of NO_x + ammonia) exceeded critical loads for avoidance of impacts across 44.5% and 85.9% of sensitive UK habitats respectively (Figure 1 a, b). The greatest excess acidity occurs in regions of Northern, Central, Eastern and South West England, as well as areas of Wales, southern Scotland, and Northern Ireland (Figure 1a). Exceedance of critical loads for nutrient N remains widespread across the UK, with only the far North of Scotland being free from excess N (Figure 1b). The area of sensitive habitat exposed to excess acidity, as well as the magnitude of exceedance, is gradually declining, reflecting reduced emissions of NO_x and SO_2 (Rowe et al., 2023). This latest assessment also continues a downward trend in nutrient N exceedance, equivalent to 7.9% or 3kg N ha^{-1} since 2002-2004 and driven mostly by reduced deposition in Scotland (Rowe et al., 2023).

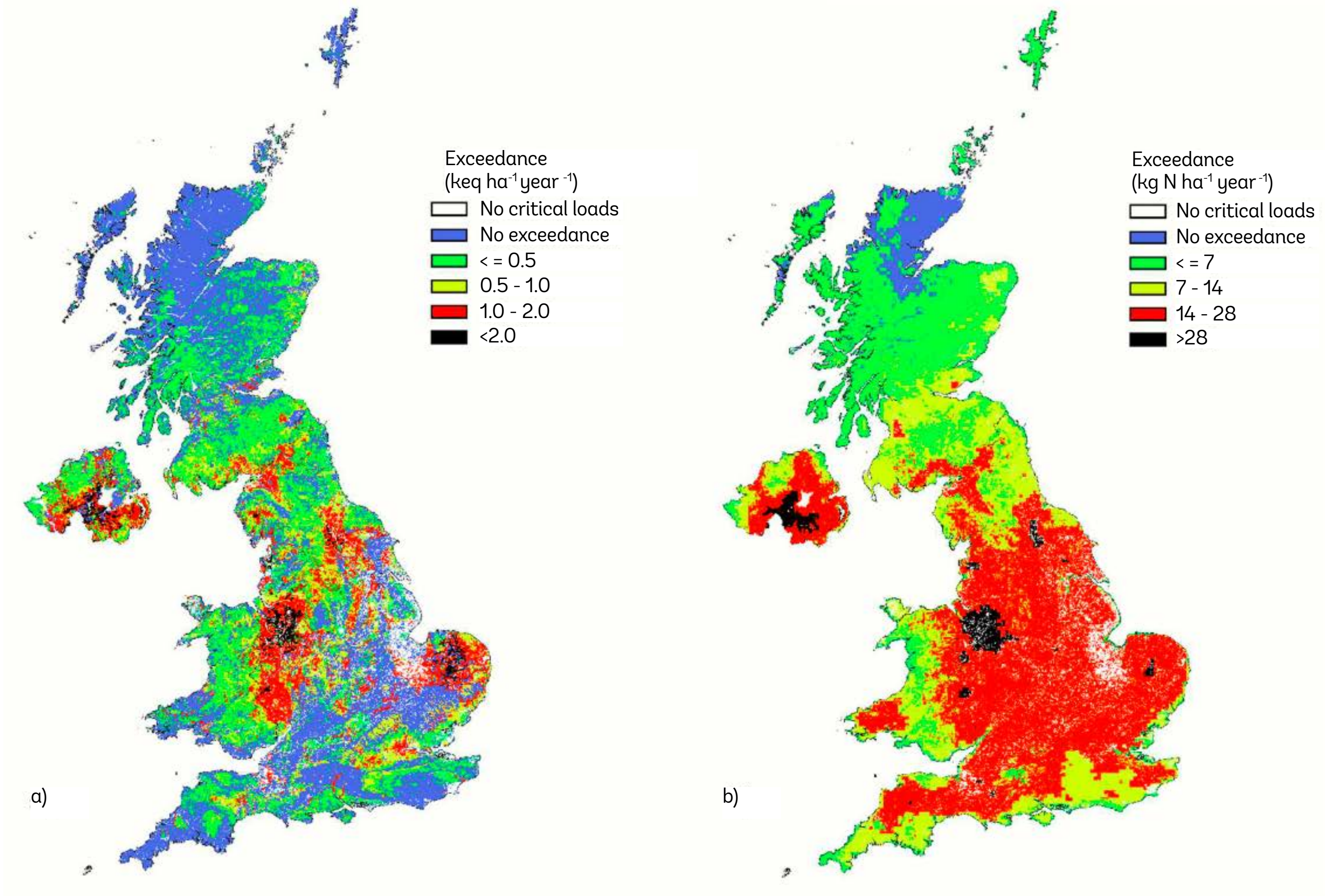


Figure 1: Average accumulated exceedance of a) acidity and b) nutrient N above critical loads for sensitive habitats in 2020, the middle year for the 2019-2021 data period (adapted from Rowe et al., 2023)

Nutrient N deposition for 2019-2021 exceeded critical loads for 88% of managed conifer woodlands, and over 90% of mixed broadleaved, beech, acidophilus (oceanic) oak, Scots pine or mixed woodlands across the UK (Rowe et al., 2023) (Table 2). There has been little change in the extent of deposition across these habitats on a UK-wide (Table 2) or regional basis since 2003 (Table 2). However, the magnitude of exceedance for these habitats is gradually decreasing across the UK (Table 2), apart from broadleaved, acidophilous oak and mixed woodland habitats in Northern Ireland (Table 2) (Rowe et al., 2023). The area of coniferous or broadleaved woodland with excess acidity is also decreasing across every country of the UK (Rowe et al., 2023). Nevertheless, acidifying pollution and nutrient N deposition continue to exceed critical loads for designated areas of global conservation importance; including Sites/Areas of Special Scientific Interest (SSSIs/ASSIs), Special Areas of Conservation (SACs) and Special Protection Areas (SPAs) (Rowe et al., 2023).

Table 2: Estimated percentage area (%) of N-sensitive woodland habitats in the UK and home nations where critical N loads are exceeded, and magnitude of exceedance in kg N ha^{-1} yr^{-1} for 2002-2004 and 2019-2021 (by convention referred to as 2003 and 2020) (adapted from Rowe et al., 2023)

	year	Managed coniferous	Broadleaved	Beech	Acidophilous Oak	Scots Pine	Mixed
UK	2003	91.4 (13.5)	97.4 (20.1)	100.0 (20.9)	96.3 (16.4)	100.0 (9.4)	97.9 (17.6)
	2020	88.0 (9.7)	96.7 (15.9)	100.0 (15.6)	92.0 (12.0)	100.0 (6.0)	94.5 (14.9)
England	2003	100.0 (22.0)	100.0 (23.5)	100.0 (21.4)	100 (22.0)	NA	100 (22.8)
	2020	100.0 (17.1)	100.0 (18.2)	100.0 (15.8)	100 (16.3)	NA	100 (17.7)
Wales	2003	100.0 (19.8)	100.0 (17.4)	100.0 (18.3)	100 (17.7)	NA	100 (18.2)
	2020	100.0 (13.6)	100.0 (13.6)	100.0 (14.3)	100 (13.5)	NA	100 (14.1)
Scotland	2003	86.0 (8.8)	87.9 (9.5)	100 (14.5)	87.4 (6.5)	100 (9.4)	91.1 (8.7)
	2020	80.4 (5.7)	84.4 (7.2)	100 (14.0)	72.7 (3.6)	100 (6.0)	76.8 (6.3)
Northern Ireland	2003	100 (15.7)	100 (20.6)	NA	100 (16.0)	NA	100 (17.8)
	2020	100 (15.4)	100 (21.5)	NA	100 (16.5)	NA	100 (18.3)

In 2019-2021, the annual atmospheric concentration of NH_3 exceeded a critical level of 1μg m^{-3} for risks of impacts on sensitive lichens and bryophytes across 64.3% of the UK land area, an increase from 56.7% in 2002-2004 (Figure 3) (Rowe et al., 2023). A critical level of 3μg m^{-3} for impacts on sensitive vascular plants was also exceeded in agriculturally-intensive regions of Northern Ireland, England and Wales (Figure 3). Average exposure to NH_3 is rising for sensitive woodland habitats and designated areas on a UK-wide (Table 3) and regional (Rowe et al., 2023) basis.

Table 3: Estimated percentage area (%) of woodland habitats and designated areas including woodland and forest sites in the UK where a critical level of 1μg m^{-3} was exceeded for 2002-2004 and 2019-2021 (adapted from Rowe et al., 2023)

UK Woodland Habitat/ designated area	2002-2004	2019-2021
Managed coniferous	21.0	29.2
Broadleaved	77.4	84.2
Beech	74.6	92.9
Acidophilous oak	48.1	62.1
Scots pine	0.9	1.6
Mixed	65.8	78.1
SSSI/ASSI	62.7	71.9
SPA	41.1	46.4
SAC	49.3	59.8

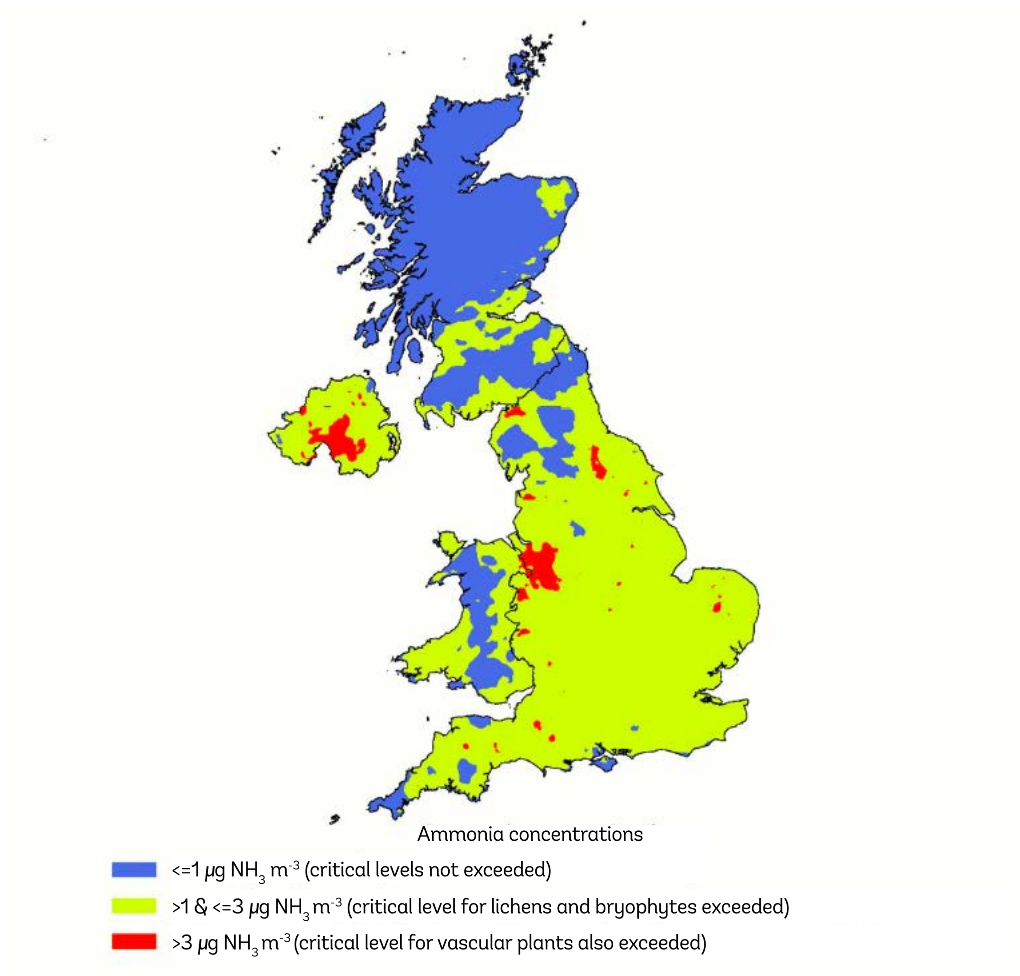


Figure 3: Estimated annual mean atmospheric concentrations of ammonia across the UK for 2019-2021 (after Rowe et al., 2023).

Data from the long-term International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects (ICP) Forest Monitoring Network sites across the UK suggests decreasing trends in the bulk precipitation and canopy throughfall of nitrate (NO_3^-) and ammonium (NH_4^+) for upland or lower deposition areas since the mid-1990s, supporting a gradual reduction in overall N pollution (Vanguelova et al., 2023). However, for sites in areas with high N deposition or nearby pollution sources, trends in NO_3^- and NH_4^+ are stable (Vanguelova et al., 2023). An increase in dissolved organic nitrogen (DON), has also been observed at some UK forest sites, the cause of which is uncertain but may reflect seasonal differences between N species or transboundary pollution (Vanguelova et al., 2023).

Current impacts

On N-limited soils, increased growth of some species may be seen in shelterbelts close to nearby pollution sources or at forest edges where N can be scavenged by tree canopies; including height, diameter and leaf area index (Vanguelova et al., 2023). Although N deposition is expected to have enhanced the growth of trees across the UK, a clear pattern is difficult to detect nationally due to confounding factors, exceedance of photosynthetic thresholds and non-linear responses of forest ecosystems (Etzold et al., 2020; Vanguelova et al., 2023). N enrichment may also influence the frequency of fruiting of plants and fungi, which may have long-term implications for regeneration within temperate woods (Van der Linde et al., 2018; Vanguelova et al., 2023). There is no strong evidence that excess N is directly damaging the crown condition of native broadleaf species; however, effects such as increased foliar N and altered root architecture may be increasing the susceptibility of sensitive broadleaf species to pest or disease and abiotic stressors such as drought (Vanguelova et al., 2023; Dietrich et al., 2024). As well as warmer temperatures and lower water availability, the increasing presence of acute oak decline (AOD) across England and Wales is positively associated with dry deposition of N, supporting this hypothesis (Brown et al., 2018; Vanguelova et al., 2023) (Figure 4). However, the mechanism of how N enrichment (and other factors) can influence AOD has not been explained (Gosling et al., 2024). Nutritional deficiency has now been identified at sites in Europe and the UK due to increased nutrient demands (Jonard et al., 2014; Vanguelova and Pitman, 2019; Vanguelova et al., 2023).

Conifer species, including native Scots pine (*Pinus sylvestris*), are generally more sensitive to excess deposition, and monitoring suggests an increased likelihood of negative impacts on crown condition, root or stem growth, micronutrient deficiencies, and susceptibility to stress and disease (Vanguelova et al., 2023). Due to differences in soil type and nutrient cycling, conifer woodlands in the UK are also more likely to experience N saturation, with increased leaching of NO_3^- into water courses and enhanced emissions of N_2O, NO and CH_4 during disturbances, such as soil warming, clear-felling and PAWS restoration (Vanguelova et al., 2023).

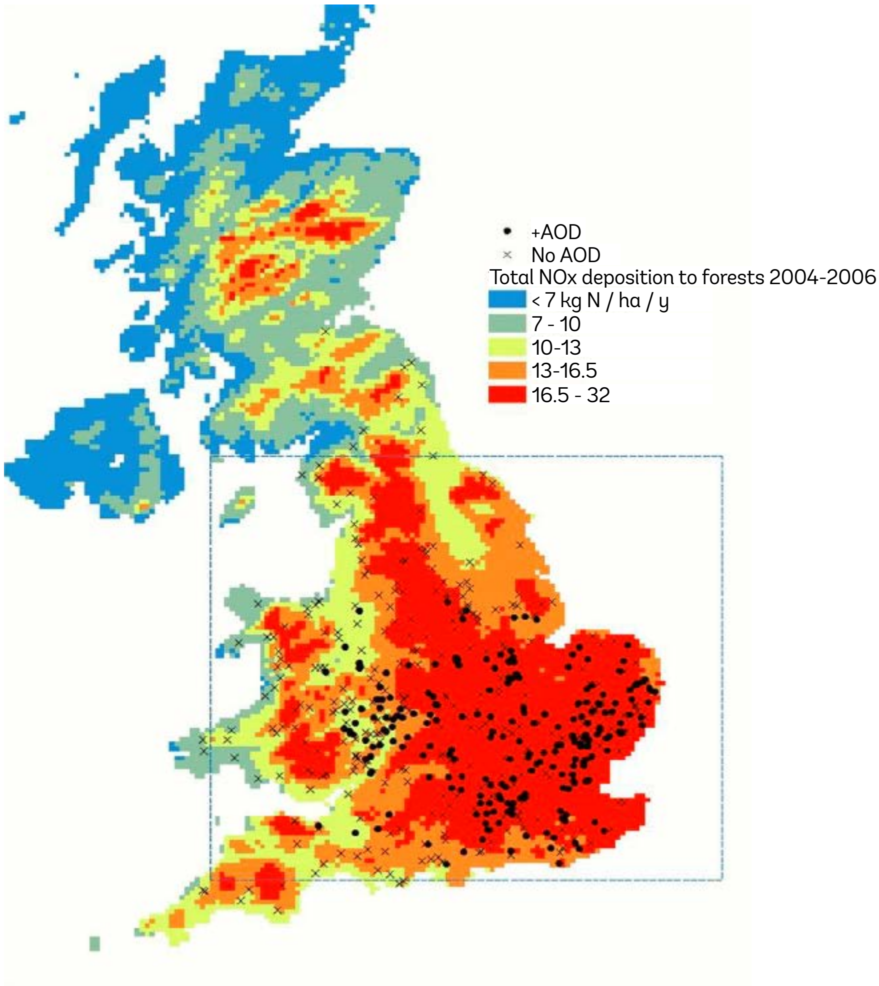


Figure 4: Map of modelled N deposition for 2004-2006 and the presence of Acute Oak Decline (AOD) across the UK (5 x5 km grid). Black dots represent sites with acute oak decline syndrome and grey dots represent healthy oak forests (after Vanguelova et al., 2023).

The deposition of N compounds or their reaction products alters the pH and chemistry of leaf, bark and soil surfaces, and is known to cause associated effects, such as increased mobility of toxic heavy metals at forest sites (e.g. Nisbet & Evans, 2014). Although recovery of vegetation from the acidifying effects of S pollution is widely indicated (Tipping et al., 2021), the effects of air pollutants on pH on a site level are cryptic and subject to biological lags, making attribution of impacts difficult (APIS, 2016a). The Bunce woodland survey, monitoring 50 years of change (1971-2021) in broadleaf woodland plots across the UK, found an overall increase in soil pH, suggesting some recovery from historical S pollution. However, a decreasing trend from 2000 to the present (Smart et al., 2024) may reflect the effect of increasing NH_3 concentrations on soil acidity (Warner et al., 2017; Rowe et al., 2023). The Bunce survey was unable to directly attribute changes in soil pH to cumulative S or N deposition (Smart et al., 2024).

The effects of N deposition on woodland biodiversity are complex, and mediated by a range of factors, including soil type, tree age and size, management, edge area and previous land use (Vanguelova et al., 2023). Annual exceedance of critical loads for acidity and nutrient-N in sensitive woodland habitats (Table 2) (Rowe et al., 2023) indicates an ongoing risk to species composition. Long-term recording data shows a disproportionate decline in the abundance of plants adapted to low N conditions or intermediate pH across Great Britain (Stroh et al., 2023). As an indicator of site soil fertility, the Bunce woodland survey also found a significant increase in mean Ellenberg N index, confirming a shift towards species adapted to more fertile conditions (Smart et al., 2024) (Figure 5). As with soil pH, a statistical impact of cumulative N deposition on Ellenberg N values was not detected in the Bunce survey, suggesting that the long-term effects of N pollution are obscured by changes to canopy and shading, as well as a lack of management (Smart et al., 2024). Similar results have also been observed at monitoring sites across Europe (Vanguelova et al., 2023). At a site level, UK woodlands are recorded responding to NH_3 released as point-source pollution, such as from intensive poultry units (IPU), with a distance-dependent decline in understory species richness, and an increased presence of algae, grass, shrubs and ferns and dominant nitrophilous species such as nettle and bramble (Vanguelova et al., 2023). Woodland Trust analysis of IPU permissions granted between 1992-2022 within the central Welsh border regions found that 487 of 1,124 approved sheds were within 500m of ancient woodland (Figure 6a), and that over 4,800ha of ancient woodland habitat is within 1km of an IPU (Figure 6b), indicating a potential risk to rare vegetation communities. Further expansion of IPUs since 2022 is known across this region, and similar patterns are expected across other agriculturally-intensive regions, such as Lincolnshire (A. Caffyn, pers. comm).

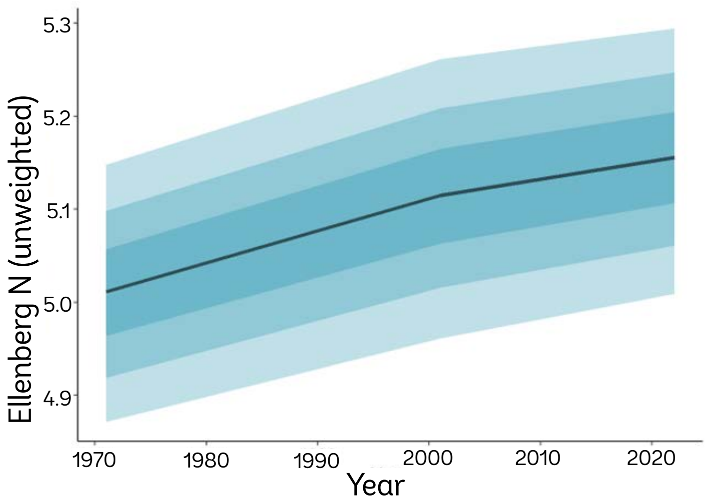


Figure 5: Increase in the mean Ellenberg N score of broadleaf woodland plots across the UK, as part of the Bunce Woodland Survey (1971-2021) (blue areas are confidence intervals, after Smart et al., 2024)

Changes in UK woodlands are also thought to include shifts in the composition of soil fungi and microbiota, with specialised ectomycorrhiza communities in conifer woodlands (e.g. *Piloderma* spp.) and users of organic N such as webcaps (*Cortinarius* spp.) particularly sensitive to N deposition (van der Linde et al., 2018; Vanguelova et al., 2023). Gradient studies of soil cores taken from ICP monitoring sites across Europe and the UK suggests canopy throughfall depositions above 5.8kg $ha^{-1} yr^{-1}$ and 9.2kg $ha^{-1} yr^{-1}$ are sufficient to induce shifts in ectomycorrhizal communities in conifer and deciduous woodlands respectively, indicating that widespread changes have already taken place (Suz et al., 2014; Van der Linde et al., 2018; Suz et al., 2021). Change to the composition or activity of soil microbiota, including bacteria, may also influence a range of processes including N-fixation, decomposition and nitrification (Zechmeister-Boltenstern et al., 2011; Bobbink et al., 2022). Occupancy trends for bryophytes and lichens suggest shifts towards species with high N requirements or tolerance of alkaline conditions due to increased environmental exposure to NH_4^+ (e.g. Pakeman et al., 2022). Contrasting patterns have also been seen for these groups in Scotland, with relative declines in the abundance of woodland lichens, and increases in bryophytes (Pakeman et al., 2022). This may indicate a growing competitive disadvantage for many acidophilous lichen species, contending with increasing fragmentation and genetic isolation, long-lasting change to soil or bark pH, and decreasing light levels (Hofmeister et al., 2022; Pakeman et al., 2022). Rising NH_3 exposure (Table 3) indicates a loss of N-sensitive lichens across the UK, such as tree lungwort (***Loberia pulmonaria***), extinct at sites across central and eastern England (British Lichen Society, 2024a). The ongoing recovery of S-sensitive species, such as string-of-sausage lichen (***Usnea articulata***) has been reported (British Lichen Society, 2024b)

Table 3: Estimated percentage area (%) of woodland habitats and designated areas including woodland and forest sites in the UK where a critical level of 1µg m^{-3} was exceeded for 2002-2004 and 2019-2021 (adapted from Rowe et al., 2023)

UK woodland habitat/designated area	2002-2004	2019-2021
Managed coniferous	21.0	29.2
Broadleaved	77.4	84.2
Beech	74.6	92.9
Acidophilous oak	48.1	62.1
Scots pine	0.9	1.6
Mixed	65.8	78.1
SSSI/ASSI	62.7	71.9
SPA	41.1	46.4
SAC	49.3	59.8

The effects of N deposition on UK woodland fauna are under-investigated and likely to be highly species-specific, and critical load thresholds for negative impacts have not been identified (Bobbink et al., 2022). However, evidence from temperate forests suggest elevated concentrations of toxic compounds such as NHO3 and NH4+, will negatively impact a range of environmental indicator groups; including springtails, earthworms, arachnids and molluscs (Bobbink et al., 2022, and references therein). Increased concentrations of N in plant tissues may enhance the feeding rate and population of nitrophilous species such as beech aphid (Phyllaphis fagi) (Bobbink et al., 2022). N-induced changes to plant and soil health as well as woodland structure may be having wide-ranging effects on higher trophic levels. For example, increased density of understorey vegetation is implicated in the ongoing decline of woodland specialists across the UK, such as wood warbler (***Phylloscopus sibilatrix***), which favour lower ground cover (Mallord et al., 2012; Stanbury et al., 2021). Habitat acidification increases heavy metal bioaccumulation and physiological stress in birds and may significantly reduce reproduce success (Sanderfoot and Holloway, 2017). Conversely, in some contexts the productivity benefits associated with air pollution may be increasing food availability and lead to higher broods (e.g. in great tits (*Parus major*) (Costa et al., 2011)). The chronic effects of NH_3 or NO_x inhalation from nearby pollution sources are unknown. However ongoing impacts on health markers of woodland birds and other vertebrates cannot be ruled out (Barton et al., 2023).

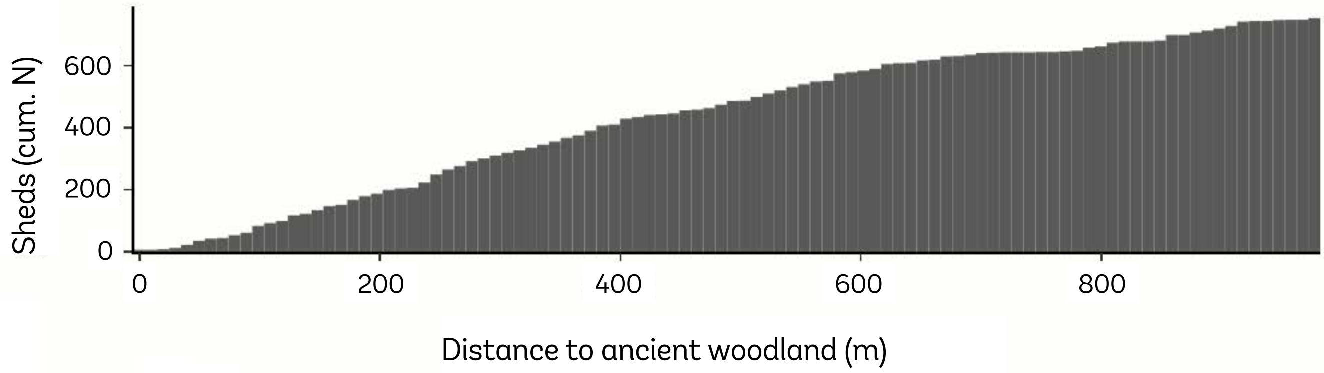
Future impacts

Total N deposition has likely peaked across Europe, with a decreasing trend in nutrient N exceedance. However, the cumulative N burden, existing levels of deposition and rising NH_3 concentrations will affect UK woodland for the foreseeable future. N deposition has already caused lasting change in UK woodlands, with shifts in the composition of vegetation and soil microbiota indicated on a national scale (Suz et al., 2014; Smart et al., 2024). Additional deposition risks N saturation in the most heavily polluted regions, and tipping points like widespread phosphorous limitation may eventually be reached with irreversible shifts in woodland condition, composition or function (Suz et al., 2021). Emerging evidence indicates that N deposition, in combination with other factors, can increase the susceptibility of some species to stress and disease, or reduce their resilience to climate change (Brown et al., 2018; Dietrich et al., 2024). The high level of fragmentation of UK woodlands is a particular risk factor in enhancing their susceptibility to the effects of N deposition. However, many impacts of N pollution are cryptic or non-linear. A lack of monitoring hinders a more complete understanding of current impacts, as well as the development of suitable critical load thresholds for temperate woodlands (Bobbink et al., 2022). The acute sensitivity of soil fungi, lichens and lower plants in boreal forests, where change to community composition may occur at a deposition load of just 2-5 kg N ha^{-1} $year^{-1}$, implies that current levels of N pollution are likely well above the level needed for the most sensitive woodland species or processes to recover (Bobbink et al., 2022). Conversely, excess N to date has been broadly positive for the growth and health of broadleaf trees (Vanguelova et al., 2023). This variation in sensitivity presents a challenge to meaningfully protect UK woodlands from N pollution and predict future outcomes.

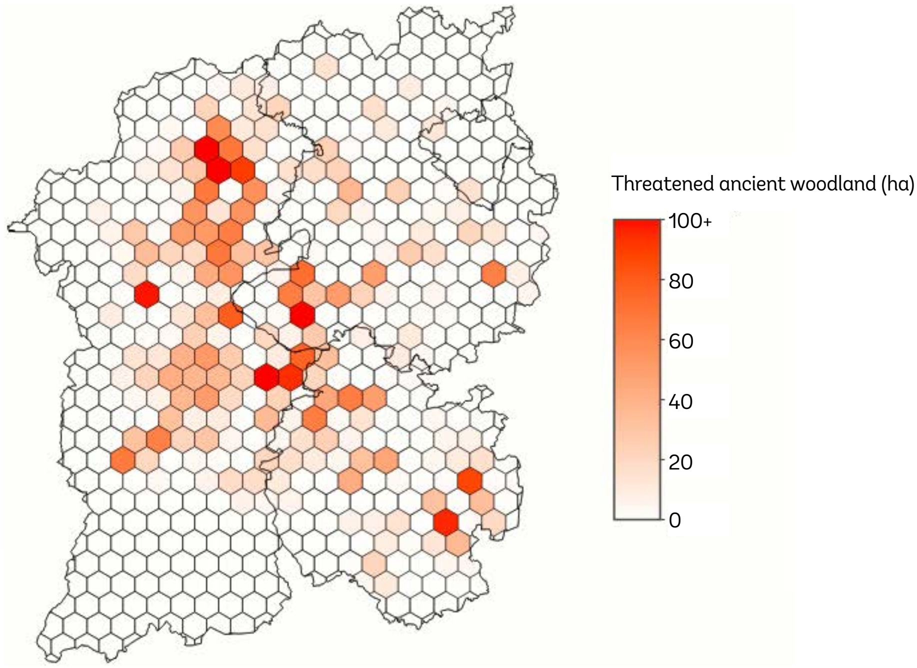
Ground level ozone (O_3)

Sources and current trends

Tropospheric (or 'ground level') ozone (O_3) is a secondary pollutant formed from precursor compounds reacting in the presence of sunlight. Levels of O_3 are thus intrinsically related to the concentration of other air pollutants and are closely determined by the volatile organic compound (VOC) to NOx mixing ratio (Mazzuca et al., 2016). Concentrations of O_3 are higher in rural, upland and remote areas where there are large amounts of natural VOCs and are lower in urban areas due to chemical depletion by NO_x (Guirreiro et al., 2014). Background levels of O_3 across the northern hemisphere mid-latitudes have increased steadily during the industrial era, rising from pre-industrial means of 5-15ppb to 33-50ppb (Pavelin et al., 1999; Sicard et al., 2017). Local concentrations of O_3 reach peak levels (>60ppb) during the spring or summer, where warmer temperatures and high light levels enhance the rate of production in acute O_3 episodes (Diaz et al., 2020).



a)



b)

Figure 6: a) Distance of granted IPU sheds to ancient woodland and b) sum of area of ancient woodland habitat within 1000m of an IPU across Powys, Shropshire, Hertfordshire and West Worcestershire

As well as being a greenhouse gas, O_3 is a powerful oxidant and is phytotoxic above a species-specific dose. Modelling performed by UKCEH for the EU National Emissions Ceilings Directive (NECD) (Sharps et al., 2022) in 2022, estimated that the annual dose of O_3 exceeded a critical level of 5.2mmol m^{-2} necessary for a 4% reduction in birch or beech tree biomass across the majority of mainland UK mixed broadleaf and beech woodland habitats in 2019 (Figure 6). For mixed broadleaf woodland, the magnitude of exceedance was greatest in South West England, and coastal regions of South Wales/South East England. The lowest overall exceedance was found in Scotland (Figure 6a). Exceedance for beech woodland habitat was also greatest in South East England (Figure 6b). Conifer species are less sensitive to O_3, and a critical level of 9.2mmol m^{-2} for a significant impact on managed coniferous woodland has not been exceeded in any year of NECD reporting (2014-2019) (Sharps et al., 2022).

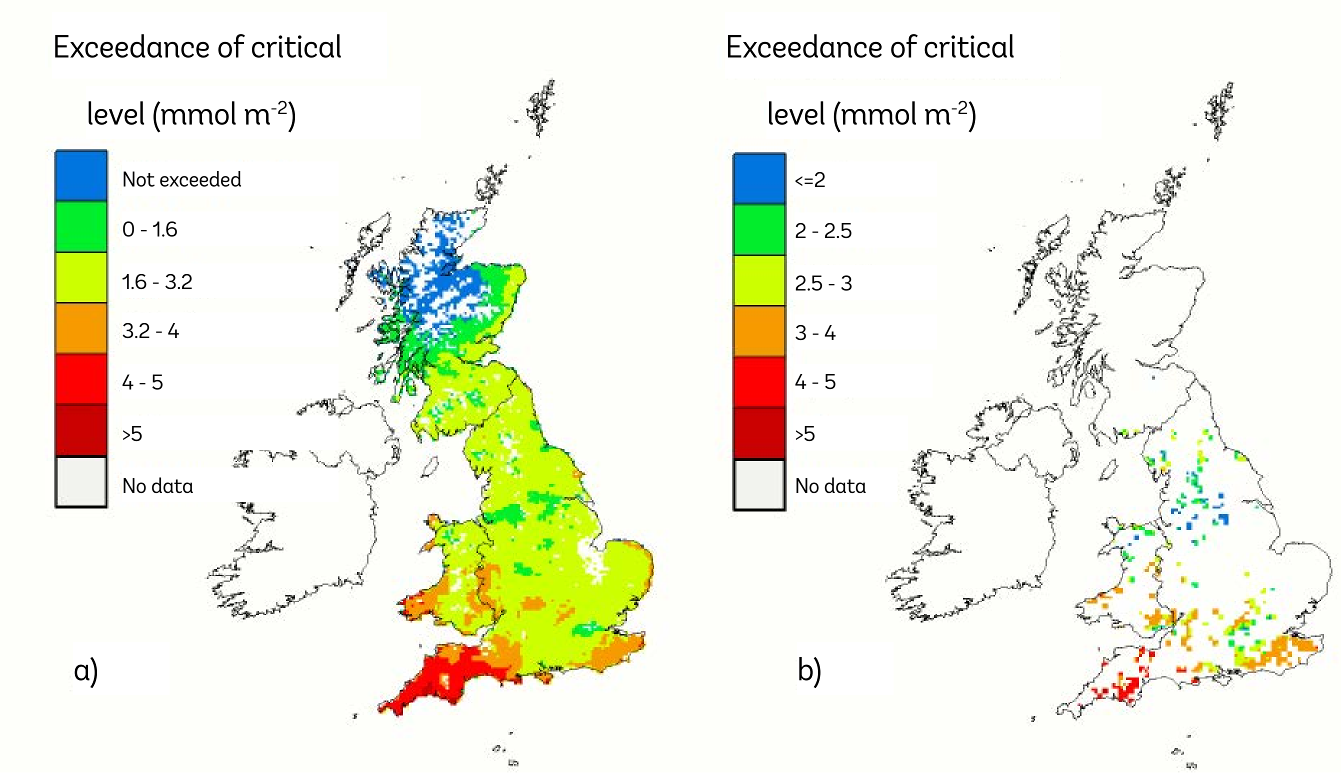


Figure 7: Estimated exceedance across the UK of the 5 mmol m^{-2} critical O_3 dose, necessary for a 4% reduction in tree biomass in a) managed broadleaf b) unmanaged beech habitats for 2019 (after Sharps et al., 2022)

Since NECD reporting began (Sharps et al., 2019) exceedance statistics for woodland habitats have been stable across the UK with no overall trend emerging, but with annual variation in the extent of ozone fluxes, biomass losses and magnitude of exceedance between years (Sharps et al., 2022).

Current impacts

In 2019, exceedance of the critical level for broadleaf trees (Figure 7) is estimated to have caused an average 6.9% loss in biomass increment for mixed broadleaf and 7.6% for beech woodland habitat across Great Britain (Table 4) (Sharps et al., 2022). Similar results are estimated on an annual basis (Sharps et al., 2019, 2022). Exceedance of critical levels provides a biologically relevant indication of the potential impact from O_3 on vegetation and woodland ecosystems, and a target for air pollution control policy. However, a lack of monitoring in UK woodlands is a limitation in developing a more comprehensive understanding of current O_3 impacts (Sharps et al., 2022).

Table 4: Estimated biomass increment loss (%) in 2019 for managed broadleaf and unmanaged beech woodland habitats across mainland UK (edited after Sharps et al., 2022).

	managed broadleaf woodland (%)	unmanaged beech woodland (%)
UK	6.9	7.6
England	7.4	7.7
Wales	7.5	7.6
Scotland	5.7	7.2

Evidence from fumigation experiments suggests O_3-induced stress may lead to system-wide changes to woodland condition and functioning similarly to other habitats (Morrissey et al., 2007; Agathokleous et al., 2020). Amongst other effects, this may include visible leaf injury and senescence in both understorey plants and trees (Figure 8), root:shoot ratios, declines in characteristic species richness, impacts on forage and litter quality, changes to flowering and fruiting, changes to soil microbe composition and activity, and altered nutrient cycling including N fixation (Aneja et al., 2007; Morrissey et al., 2007; Parsons et al., 2008; Lindroth, 2010; Agathokleous et al., 2020; Dugue et al., 2021). O_3 gas can also directly impact insect behaviour and reduce odour recognition in pollinators and parasitoids at current background levels (Gate et al., 1995; Ryalls et al., 2022) and may influence the population dynamics of birds (e.g. Reif et al., 2023). Sensitivity to O_3 varies widely between species and differential responses are likely within UK woodlands, with a relative susceptibility of communities in woodland fringes and clearings, deciduous trees, legumes, willows, light-loving species and species on dry sites (Mills et al., 2007; Agathokleous et al., 2020). Responses to O_3 are often mediated by environmental and local site factors including competition, nutrient availability, ambient CO_2, and water availability. For example, European beech (***Fagus slyvatica***) trees may experience reduced growth in response to O_3 in well-watered conditions, but enhanced growth in droughted conditions, perhaps due to the interactive effects of these stressors on stomatal function (Hayes et al., 2015). Reducing root biomass to maintain photosynthetically active shoot biomass also increases sensitivity of trees to drought and windfall events (Hayes, pers. comm).

FELICITY HAYES

Characteristic leaf injury in silver birch (Betula pendula) in an ozone fumigation experiment

Biomonitoring studies across Europe suggest the impacts of O_3 on temperate forests are complex and difficult to extrapolate from fumigation experiments, which have most often used young trees in isolated mesocosms or open-top chambers, due to the practical difficulty of experimentation. The magnitude of O_3 impacts on forest growth varies spatially and temporally and may be offset by - or co-linear with - other environmental factors such as elevated CO_2, N deposition, and drought (Lindroth, 2010; de Vries et al., 2014; Paoletti et al., 2017; Cailleret et al., 2018; Braun et al., 2022). For example, Paoletti et al. (2017), found no significant association of ambient O_3 with the strand volume growth of ***F. sylvatica*** in sites across Italy, whereas Braun et al., (2022), found good agreement between experimental and epidemiological data in Swiss populations. Quantifying the impact of O_3 on UK woods and trees requires the integration of fumigation experiments, an expansion of high-frequency biomonitoring and multivariate analysis and modelling approaches (Paoletti et al., 2017; Cailleret et al., 2018; Braun et al., 2022; Sharps et al., 2022).

Future impacts
Since 2000, rural trends for O_3 have been relatively stable across the UK, whereas urban O_3 has increased slightly due to improved control of NO_x (AQEG, 2022). Future levels will reflect multiple drivers, including global concentrations of transboundary pollutants such as CH_4 as well as downward transport from the stratosphere. Depending on the socioeconomic emission pathways followed in the UK and around the world, O_3 could continue to increase or reach a peak in the UK and Europe within the next decade. However, surface concentrations are likely to remain above critical levels for impacts on broadleaf woodland for the foreseeable future (AQEG, 2022; Sharps et al., 2022).

Particulate matter

Sources and current trends
Particulate matter (PM) comprises a variety of fine-grained solid or liquid aerosols from a wide range of anthropogenic and natural sources including combustion of fossil and wood fuels, construction, road traffic, metal working, pollen, sea spray and dust storms. Concentrations and exposure of PM are typically higher in urban than rural areas and show a strong association with socio-economic deprivation (Milojevic et al., 2017). PM has significant impacts on human health both nationally and globally and is subject to controls under air quality standard regulations (DEFRA, 2024). Long-term decreases in annual mean concentrations of PM particles less than 10 (PM_{10}) or 2.5 micrometres ($PM_{2.5}$) have been recorded at roadside and urban monitoring sites across the UK, reflecting improved control of emission sources (DEFRA, 2024). Critical loads or thresholds for PM impacts in woodland ecosystems have not been developed (Bobbink et al., 2022), and national mapping of PM for woodland habitats on a UK scale, including the urban forest, is unavailable.

Current impacts
The effects of PM on woodlands are poorly understood. PM reduces the quality of light and may be removed directly from air by plants via the stomata, or deposited as dust on leaf, bark, or soil surfaces. Moss, herb, shrub and tree species are all recorded as accumulating PM with individual distance-dependant responses (e.g. Popek et al., 2022). The efficiency of trees to remove PM from the air is species-specific, but trees with dense or long-lived canopies, such as beech or spruce, are generally understood as having a higher capacity to remove PM and other pollutants from the air (Grote et al., 2016). Sensitivity to PM also varies, and plants and trees may display a variety of stress responses such as elevated antioxidant levels (Dadkhah-Aghdash et al., 2022), reduced chlorophyll and leaf water content (Chen et al., 2015) and stunted root growth (Piacentini et al., 2019). Chronic exposure to air pollution, including PM, is a general contributory factor impacting the growth or condition of urban trees (Czaja et al., 2020). However, the magnitude of this effect is unknown, and is likely to be secondary or co-linear with other urban stressors such as high soil temperature, nutrient deficiency or drought (Czaja et al., 2020). Trees and plants in rural woodlands nearby to major PM sources, such as major roads and airports, may display elevated stress markers in response to PM components (Kovátz et al., 2021), including increased insect damage (Bignal et al., 2007), but evidence for this effect across the UK is

currently lacking. PM is also likely to have some negative impact on the health or condition of animals in urban woods, particularly birds (Barton et al., 2023). Finally, many individual components of PM, such as polychlorinated biphenyls (PCBs) and heavy metals, are also known to have long-lasting ecological impacts in their own right. UK urban woods are estimated to remove 0.7Kt PM2.5 per year, worth over £67million in avoided health impacts (Jones et al., 2019). UK vegetation as a whole reduces annual PM2.5 concentrations by around 10% (Nemitz et al., 2020). However evidence suggests that at a local or city-wide level, the overall contribution of trees and vegetation to air pollutant removal is limited, and that the major role of trees is in directing the flow of polluted air masses (AQEG, 2018).

Future impacts

PM originates from a wide range of sources, making any summation of future trends difficult. If negative trends for PM continue, this may indicate a reducing impact on human health and the urban forest over time. Greater awareness of the importance of urban green space for human health and climate change resilience highlights the need for an improved understanding of PM. To improve models of urban ecosystem service provision such as I-tree Eco (i-Tree, 2024), continued research on the benefit of trees for air pollution control is needed (e.g. Gaglio et al., 2022). In some contexts, trees and green infrastructure may also exacerbate PM pollution through precursor emissions of allergenic pollen or VOCs, or by concentrating polluted air in urban street canyons; urban forestry must take these potential disservices into account (Roman et al., 2022).

Other pollutants

A range of other soil, air or water pollutants are known, or have the potential, to negatively affect UK woodland ecosystems or species, including heavy metals, persistent organic pollutants (POPs), airborne microplastics, phthalates, and plasticisers (Nam et al., 2008; RoTAP, 2012; CLRTAP, 2017; Billings et al., 2023; Forest Research, 2024; Weaver et al., in press). Both currently used synthetic pesticides (CUPs) and legacy banned restricted pesticides (BRPs) pose chronic risks to biodiversity and leave persistent residues (Geissen et al., 2021). Exposure and bioaccumulation of pesticides in small mammals is thought to be pervasive in arable landscapes (Fritsch et al., 2022). The sub-lethal impacts of neonicotinoids on bees and other non-target organisms are well recognised (Basley, 2018; Klingelhöfer et al., 2022). Herbicide drift negatively affects woodland plant species and may be a long-term stressor for woodland in agriculturally intensive regions (Gove et al., 2007). In all cases, a lack of systematic monitoring and mapping hinders a more thorough understanding of their current extent and potential influence in UK woodlands. The Woodland Trust estate (and presumably woodlands in other public or private ownership), is also increasingly affected by illegal fly tipping (Figure 8), which may reflect increasing levels of financial hardship and a lower availability of local disposal or recycling facilities. The Woodland Trust is currently phasing out the use of plastic tree protection for its estate and outreach activities and is actively involved in research to develop practical alternatives. More research is also needed to understand the effects of light and noise pollution, but these likely exert a significant impact on UK

Fly-tipping at Haddock Wood in Runcorn, Cheshire. Fly-tipping is a growing problem across the Woodland Trust estate and may reflect increasing levels of deprivation and a lower availability of disposal facilities

woodlands, including inducing earlier bud-burst (ffrench-Constant et al., 2016) and disturbing the natural behaviour of birds (Carr et al., 2021).

Conclusion

Pollution continues to exert a significant influence on UK woodland ecosystems. Known impacts from experimental or monitoring studies vary from reduced odour recognition and activity in pollinators (Ryalls et al., 2022), to shifts in species composition on a national scale (Smart et al., 2024). Evidence suggests some effects, such as changes in ground flora composition or loss of sensitive lichens, may be reversible within a few years to several decades upon the cessation of pollution inputs (Stevens, 2016; British Lichen Society, 2024b). However, other changes, such as the long-term effects of soil enrichment on sensitive ectomycorrhiza, may be effectively permanent without direct intervention, with the effects of Roman agricultural activity still apparent in soil chemistry and mycorrhizal communities after 2,000 years (Diedhou et al., 2009). A lack of seed sources or appropriate management, combined with continued exceedance of critical loads or thresholds, reduces the chance of recovery from pollution and increases the likelihood of woodlands reaching ecological tipping points; with irreversible changes to structure, condition or functioning (Jonard et al., 2014; Suz et al., 2021). Many responses of woodland ecosystems to pollution are cryptic or non-linear, and subject to biological or chemical lags, meaning the full importance of current deposition levels may not become apparent for several decades to come (APIS, 2016a; Smart et al., 2024). As well as posing direct threat to human health, the ongoing effects of pollution threaten a vast range of beneficial ecosystem services, including pollination, food production and forage, aesthetics, carbon sequestration, timber production, urban cooling, and air and water quality control (Ren et al., 2011; Tai et al., 2021; Czaja et al., 2020; Franz and Zaehle, 2021; Ryalls et al., 2022; Vanguelova et al., 2023; Nowroz et al., 2024). Valuations of the cost of pollution to UK woodland ecosystem services are limited. Reductions in overall N pollution since 1990 are thought to have resulted in a net benefit of £65m to the UK economy on an annual basis (Jones et al., 2014). The cost of O_3 damage to wood production in Italian forests ranges €31.6-57m annually (Sacchelli et al., 2021).

The fragmented nature of UK woodland habitats, which expands the canopy edge area for scavenging air pollutants, heightens the risk of negative impacts compared to other terrestrial habitat types (RoTAP, 2012; CLRTAP, 2017; Rowe et al., 2022; Vanguelova et al., 2023). Emerging evidence suggests that cumulative N deposition, O_3 and PM are significant stressors for trees, and may directly or indirectly increase the susceptibility of some species to pests or diseases, or to the effects of climate change (Hayes et al., 2015; Brown et al., 2018; Cjaza et al., 2020; Bobbink et al., 2022; Vanguelova et al., 2023; Dietrich et al., 2024). Competing or interactive effects may also be seen between pollutants, adding further complexity to their influence at an ecosystem scale (Mills et al., 2016; Franz and Zaehle, 2021). Responses to pollution vary on a species-specific level, presenting a lasting challenge in meaningfully protecting woodland ecosystems using critical load or critical threshold methodologies (Bobbink et al., 2022). In all cases, a lack of biomonitoring, experimentation or modelling hinders an understanding of current or future impacts.

Knowledge and research gaps

- Improved monitoring and mapping of a range of pollutants across UK woodland habitats.
- Standardisation of monitoring and mapping procedures/frequency.
- Species-specific and ecosystem-wide impacts of pollutants (e.g. NH_3, O_3, PM).
- Research into long-term recovery from nitrogen pollution.
- Continued revision and development of critical load thresholds for protection of woodland ecosystems.
- Research into impacts of emerging pollutants (e.g. microplastics).
- Cost associated with damage to woodland ecosystem service provision.
- Impacts of pollutants on animal species.

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The impact of grey squirrels on UK woods and trees

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Summary

- Grey squirrels can cause extensive damage to trees by stripping off the outer bark and ingesting the underlying phloem tissue.
- Here we summarise the impacts of grey squirrels on UK woods and trees from a rapid literature review, and the distribution of squirrel damage presence and severity across the UK, using data from the National Forest Inventory (NFI). The NFI is a representative survey of woodland condition for England, Scotland and Wales.
- Within England, damage severity varies regionally. Woodland squares with the highest severity of damage are found in South East England, South West England and Wales.
- For example, in nearly 50% of woods surveyed in South East England, the majority of trees that showed signs of bark stripping are likely to die due to the severity of damage.
- The severity of bark stripping depends on tree species present and age of the tree. Sycamore and beech are most susceptible, with oak, sweet chestnut, birch, ash and maples also susceptible. Trees between the ages of 10 and 40 are at higher risk of damage.
- Grey squirrels pose a probable economic loss to forestry of approximately £37 million a year in England and Wales according to the Royal Forestry Society.
- Little is known about the impact of grey squirrels on wider ecosystem services provided by woodlands and how this relates to their ecological resilience.
- Beyond their negative impacts on red squirrel populations, the effects of grey squirrels on biodiversity are poorly understood.
- The severe impacts of grey squirrel damage on broadleaf trees has significantly influenced landowners' choice of tree species for planting, preventing important native broadleaves (e.g. oak and beech) from being planted. This reduces the ability of woodlands to promote biodiversity and mitigate and adapt to climate change.
- It is likely that bark stripping by squirrels reduces woodland resilience to climate change.
- Currently, novel research is investigating how to effectively reduce grey squirrel numbers and understand mechanisms underpinning bark stripping behaviour. Novel approaches to investigating grey squirrel behaviour, ecology and impacts within UK woodlands will facilitate the development of effective and targeted management methods to improve woodland resilience and adaption to climate change.

Introduction

Introduction, distribution and impacts of grey squirrels in Britain

Since eastern grey squirrels (*Sciurus carolinensis*) were introduced to Britain, they have caused extensive damage to trees from their bark stripping behaviour. The grey squirrel is native to North America and was deliberately introduced as an ornamental species to areas of Great Britain in the early 20th century. (Long, 2003). Grey squirrel damage undermines the UK Government's ambitious afforestation targets to increase woodland area and mitigate the effects of climate change. Currently, the drivers and impacts of bark stripping are poorly understood (Nichols et al., 2016), hindering the identification of effective mitigation options.

Grey squirrels are present in Northern Ireland, and within all three countries of Great Britain (Figure 1; UKSA, 2024). However, grey squirrel range is less extensive in Scotland and mainly restricted to southern Scotland below the Highland boundary fault line.

UK red and grey squirrel distribution map 2017-2022

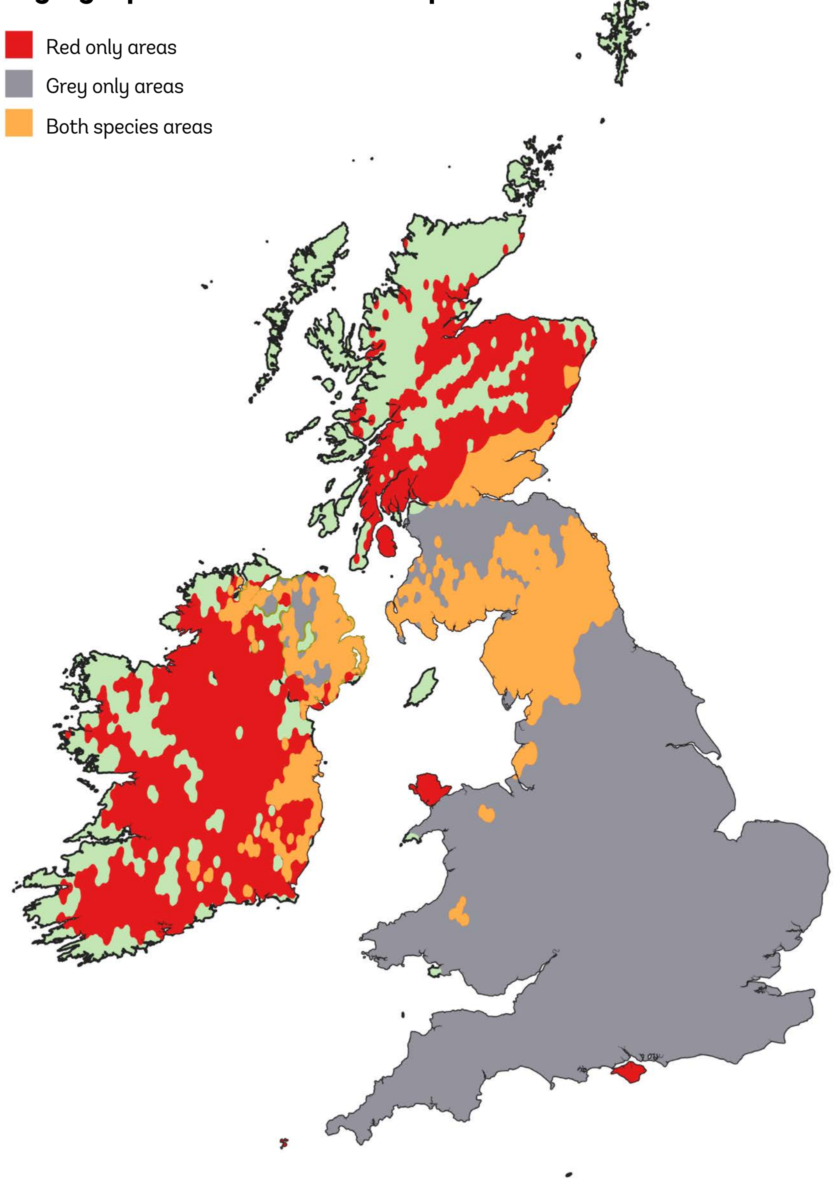

Figure 1. Courtesy of the UK Squirrel Accord (2024): Map of red and grey squirrel sightings data submitted and verified 2017-2022. Data provided by CEDaR, Clocaenog Red Squirrels Trust, Colin Lawton, Mammal Society, Mid-Wales Red Squirrels Partnership, National Biodiversity Data Centre, National Parks and Wildlife Service, Red Squirrels Northern England, Saving Scotland's Red Squirrels, Trees for Life, Ulster Wildlife, University of Galway and Vincent Wildlife Trust. It is a visual representation of the recorded presence of both species, though it may not fully reflect true populations. To produce the map, alpha hulls were used to draw polygons around the sighting records. A buffer of 5-10 km was applied to each polygon based on the requirements of the respective data providers.

Squirrel damage greatly affects timber yield and production, costing Britain up to an estimated £37 million a year (Richardson et al., 2021). While the impacts of stripping on timber product are well known amongst the forestry community (Mayle and Broome, 2013, Richardson et al., 2021), little is known about bark stripping impacts on wider ecosystem services (the ways nature supports and helps us) provided by woodlands and how this relates to their ecological resilience (how well woodland can recover after disturbance).

Bark stripping behaviour and potential causes

Grey squirrels cause extensive damage to trees across Britain by peeling off the outer bark and gnawing on the underlying phloem and cambial tissue layers (Kenward, 1983). This disrupts the flow of nutrients from the roots to the canopy of the tree and can result in structural deformities or death of affected trees (Mayle, 2004). In a study conducted in Lady Park Wood, Wales from 1977 to 2002, mortality rates were recorded between 2.3 and 5.4% per year (Mountford, 2006). Damage takes place primarily between the months of May and August, peaking in July, when tree growth is most rapid, and when squirrel densities are at their highest (Fitzgibbon, 1993, Mayle and Broome, 2013).

There appears to be spatial and temporal variations (i.e. seasonality of the behaviour, damage occurring in clusters, varying with forest composition) in bark stripping behaviour across Britain (Kenward, 1983). With the damage severity varying between landscapes and even locally within a woodland, it is difficult to allocate a single cause to the behaviour. Several causes have been proposed to explain bark stripping behaviour by invasive grey squirrels. Proposed hypotheses include exploratory feeding behaviour (Kenward and Parish, 1986, Mountford and Peterken, 1999, Bertolino, 2008), a trace nutrient deficiency (Kenward, 1983, Nichols et al., 2016), territorial marking (Taylor, 1968) and a fondness of sugar (Kenward, 1982, Kenward, 1983, Kenward, 1988). High juvenile density during late spring and early summer is consistently cited as an important factor influencing bark stripping intensity (Kenward et al., 1992, Fitzgibbon, 1993, Mountford, 2006, Mayle and Broome, 2013). This may be explained by increased agonistic behaviour with higher juvenile densities. Agonistic behaviour describes an aggressive encounter between a dominant and subordinate individual, driving the subordinate individual to strip or gnaw bark to relieve stress from the interaction (Taylor, 1966, Kenward, 1983). However, multiple proposed hypotheses could be simultaneously true, even within one woodland. There is likely to be more than one motivation for the behaviour, and these may vary in different contexts and environments.

Aim and objectives

Here we aim to summarise the geographic variation in grey squirrel bark stripping as well as the impacts of grey squirrels on UK woods and trees. We used data from the National Forest Inventory and information collected from a rapid narrative review to answer the following questions:

1. What is the current distribution and severity of bark stripping by grey squirrels across the UK?
2. What impact do grey squirrels have on the resilience of UK woods and trees?

Methods

Distribution and severity of grey squirrel bark stripping

Bark stripping damage is widely reported and is monitored locally. However, there is no national initiative to effectively evaluate the scale and severity of the problem. Forest Research's National Forest Inventory offers useful insight into the distribution of squirrel damage and contributing factors to risk. However, the NFI was not designed as a comprehensive grey squirrel bark stripping survey, and flaws in the interpretation of the data are discussed below. A report by Peden et al. (2020) is the most up to date analysis of this data and selected results are presented here. Methods for aggregating squirrel damage presence to 10km grid squares can be found in the methodology section of Peden et al. (2020).

Field data collected as part of the National Forest Inventory (NFI) incorporates over 15,000 one hectare woodland 'squares' across England, Scotland and Wales, from which data describing the site's biophysical attributes, including signs of bark stripping damage are collected using a standardised protocol. Squares were selected using a stratified-random strategy to ensure that each site contained at least some woodland, and that woodlands of different types (e.g. broadleaf or conifer) and ownership types (privately or publicly owned) were proportionally representative of their regional availability (Forest Research, 2019). Only the first wave of data collected between 2010 and 2015 was available for analysis. Damage severity and frequency was grouped to 'NUTS1' regions of the UK which divides England, Scotland and Wales into 11 geographic regions. NFI field surveyors recorded signs of squirrel damage on trees such as bark stripping. Reports of stripping damage below 1.8 metres are omitted, to remove incidences of damage that could have been caused by other mammals such as deer, rabbit and vole. As squirrel density data is poor, we are currently unable to provide explanations for bark stripping prevalence and intensity as a function of population density.

Impact of grey squirrels on the resilience of UK woods and trees

A rapid, narrative review was conducted to answer questions on damage distribution and severity as well as the impact of grey squirrels on UK woods and trees. We sought both published and grey literature from literature databases such as Google Scholar, Web of Science and Scopus.

Diverse sources of evidence were included in the review, with priority given to peer-reviewed research in academic journals, followed by book chapters and grey literature, including practical advice and guidance notes from government agencies and non-governmental organisations. No restrictions on date of publication were used when searching the literature. Studies focused on the UK were prioritised, with those focused on North America or Europe deemed relevant where appropriate to grey squirrels and their impacts, particularly where UK studies are lacking.

Evidence from the literature (or lack thereof) of the impact of grey squirrel damage was organised and summarised into the following categories:

- Economic – the monetary value of trees of woods.
- Wildlife – the value to wildlife of woods and trees, including biodiversity and nature recovery.
- Woodland Ecological Condition – indicators which are considered to represent good ecological functioning and suitability for wildlife.
- Climate change mitigation – the ability of woods and trees to sequester and store carbon from the atmosphere.

- Climate change adaptation – the ability of woods and trees, as individuals and as habitats, to adjust to the effects of both current and future climate change.
- Air quality – the ability of woods and trees to influence the composition and dynamics of air.
- Water quality and flood management – the ability of woods and trees to influence the composition and dynamics of water, both in terms of movements at a catchment scale, and the ecosystem services provided by the impacts of trees on water such as flood risk management.
- Recreation, access and wellbeing – the impacts woods and trees can have on human health and wellbeing and the ability of humans to gain access to those impacts.
- Landscape character – the features of a landscape that give it a sense of place.

Results and discussion

Distribution of grey squirrel bark stripping

The National Forest Inventory (NFI) shows evidence of bark stripping damage in woodlands within England (16% of randomly selected one hectare woodland squares with damage), Wales (11% with damage) and Scotland (0.6% with damage), see Figure 2 (Peden et al., 2020). Data on bark stripping damage is not available for Northern Ireland. Woodland in Great Britain as a whole showed 26% of randomly selected one hectare woodland squares with damage. Squirrel damage occurrence varies within countries. In South West England (32% of randomly selected one hectare woodland square with damage), Yorkshire and the Humber (23% with damage), and South East England (17% with damage) damage is more prevalent compared to the national average for England (Peden et al., 2020). However, due to limitations in the NFI's sampling approach (discussed below), these percentages are likely to be an underestimate. Figure 2 indicates that bark stripping damage can occur further north in Scotland than the estimated grey squirrel distribution in Figure 1. It may be that damage in more northerly parts of Scotland has been committed by red squirrels (the NFI did not distinguish between red and grey damage in the first wave of surveys).

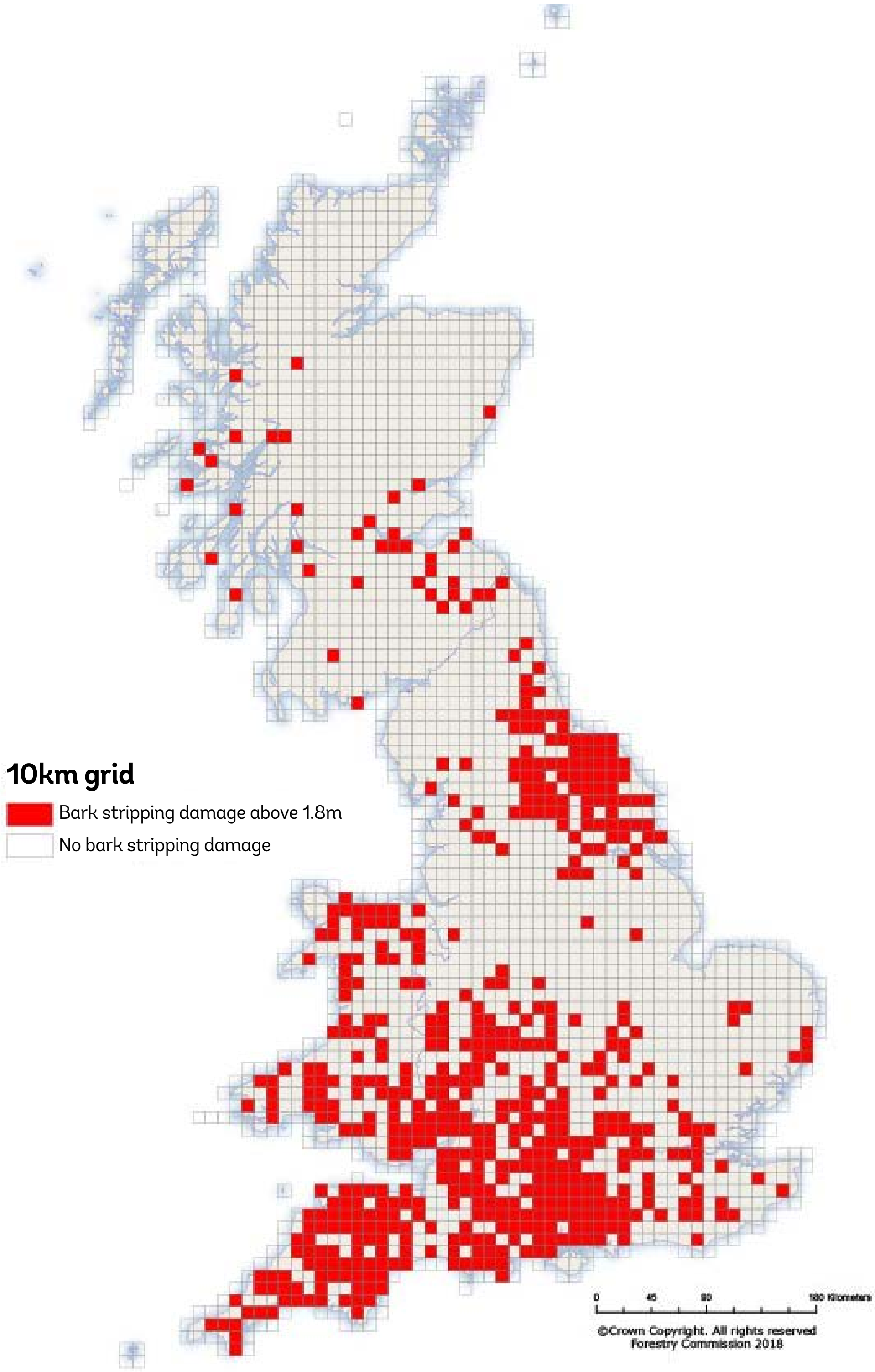

Figure 2. Locations where evidence of squirrel bark stripping was observed in the 2010-2015 NFI survey from Peden et al. (2020)

Severity and frequency of grey squirrel bark stripping damage

Like prevalence of damage, the severity and frequency of bark stripping also varies geographically. Severity of damage is determined by whether the majority of trees damaged are likely to survive or die as a result of the damage caused by squirrels, and frequency is indicated by the percentage of trees damaged within the woodland. South East England, South West England and Wales all experience the highest frequency and severity of damage (see Figure

3, Figure 4). In 49% of NFI sections surveyed in the South East of England, the majority of trees that showed signs of bark stripping are likely to die due to the severity of damage (see Figure 4). This percentage is 35% in the South West of England and 26% in Wales. Patterns in prevalence and severity of squirrel damage across the UK could be due to geographic variation in tree species presence, squirrel density, food availability, woodland cover and connectivity.

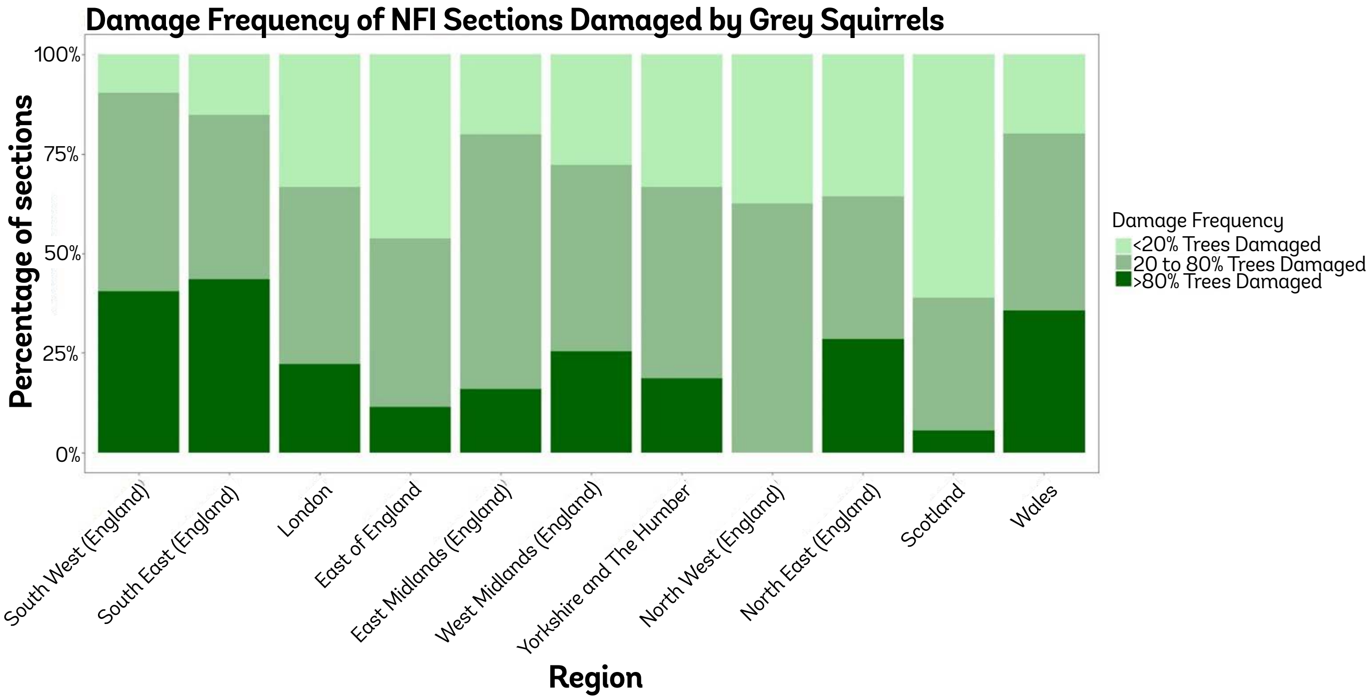

Figure 3. Damage frequency of trees within NFI sections by region

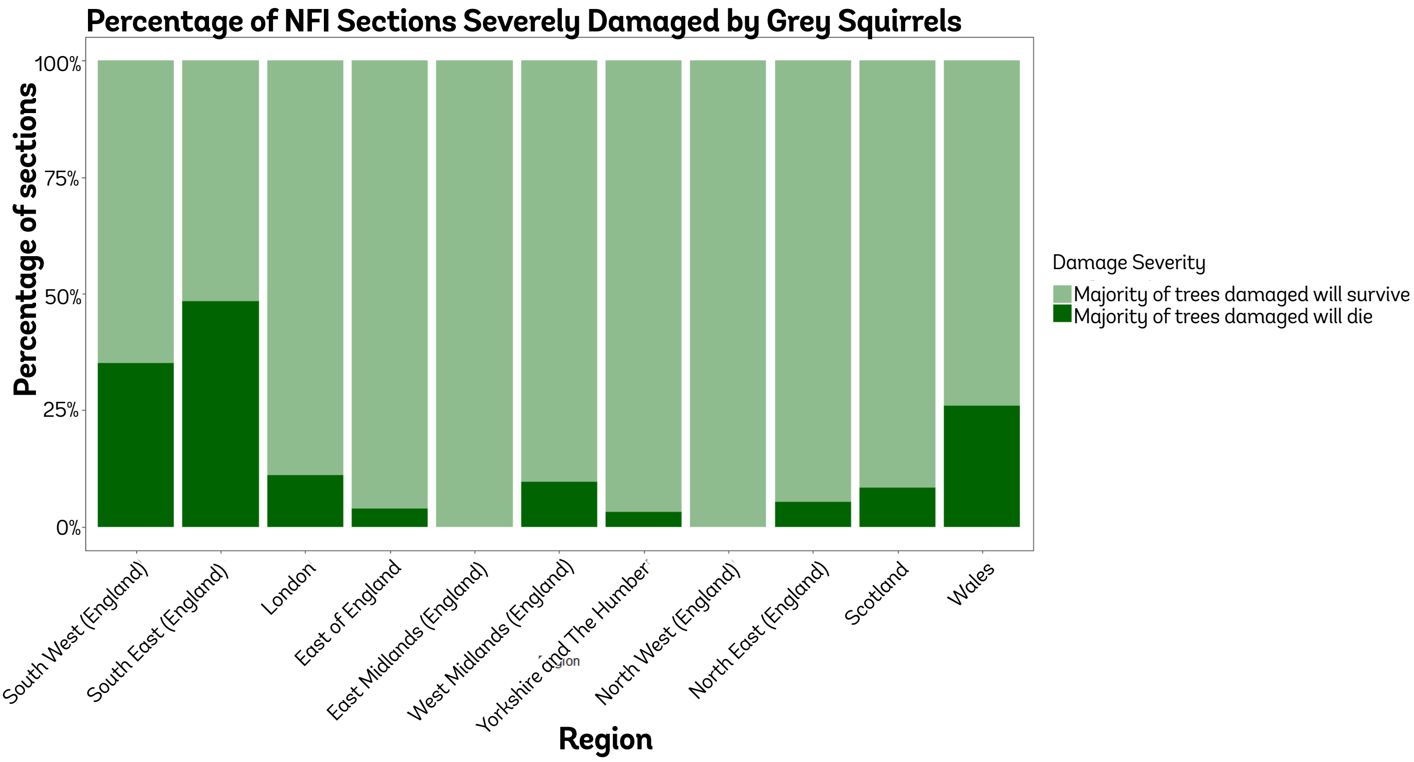

Figure 4. Percentage of NFI sections with damage where the majority of trees will die due to damage, by region

It is known that squirrels are selective in the trees they damage, making tree-level attributes such as age, size, species and location of trees within a stand important in predicting damage occurrence (Fitzgibbon, 1993, Sullivan et al., 1994, Kenward et al., 1996, Mayle et al., 2009). The NFI sampling approach is useful in assessing overall woodland susceptibility to squirrel damage, however is limited in providing information on individual tree susceptibility. The NFI aims to provide information on overall condition of UK forests and woodlands, making it essential in developing policy and guidance to sustainably manage woodland habitat. Individual tree health is not an objective of the NFI, and so makes the data limited in providing information on individual tree susceptibility to squirrel damage. The NFI records all mammal damage observed on trees, however it categorises damage by above or below 1.8 metres on the tree. Reports of damage found below 1.8 metres on the tree are omitted, as this could be caused by other mammalian species. As such, this will underestimate the true prevalence of grey squirrel damage in British woodlands. Methods to assess squirrel damage are still being developed and are being improved constantly. Squirrels can strip bark from any position on the stem and canopy of a tree. It is challenging to develop a method that allows us to observe squirrel damage at all positions on the tree, due to logistical difficulties with accessing the canopy. There is currently no practical method to access the canopy of trees when assessing squirrel damage, and this may be improved over time with new technological advances.

Despite fundamental challenges in observing squirrel damage, regional data from the NFI indicates that grey squirrel damage is widespread. Geographic variation of bark stripping prevalence is thought to be due to the proportion of favoured tree species across England, such as sycamore and beech (Peden et al., 2020). There is some debate regarding the rank order of broadleaf tree species that are most susceptible to squirrel damage (Richardson et al., 2021). However, it is widely accepted that sycamore and beech are considered highly susceptible to bark stripping damage, with oak, sweet chestnut, birch, ash and maples also very susceptible (Rowe and Gill, 1985; Mayle et al., 2013). This categorisation is confirmed by the 2010-2015 NFI field cycle data (Peden et al., 2020) and is likely to have remained largely unchanged over the last 50 years (Rowe and Gill, 1985). These species are all classed as among the most common tree species found in broadleaf woods (Smart et al., 2024), indicating the scale of the problem.

It is generally thought that trees between the ages of 10 and 40 are at a higher risk of damage on the trunk (Middleton, 1931, Shorten, 1957, Rowe, 1967, Rowe, 1984, Rowe and Gill, 1985, Gurnell and Pepper, 1988, Gurnell, 1989, Kenward and Dutton, 1996, Rayden and Savill, 2004). It has also been shown that trees up to 80 years old are at high risk of bark stripping, but younger trees (<20 years) are more likely to die as a result of the damage (Peden et al., 2020). It is also worth noting that whilst mature trees over the age of 40 are thought to be less susceptible to squirrel damage, bark stripping in these trees often occurs in the canopy, which is challenging to observe. Tree size and dominance may be a better predictor of risk than age, with trees of between 7.5 and 35cm diameter at breast height being most at risk of damage on the main stem (Mayle and Broome 2013). Figure 5 indicates that woodlands with a higher age range of trees present are more likely to show signs of damage. This may be due to the presence of both younger trees at higher risk of damage, and older trees providing food resources to maintain a high squirrel population.

Age range of trees damaged by squirrels across Britain

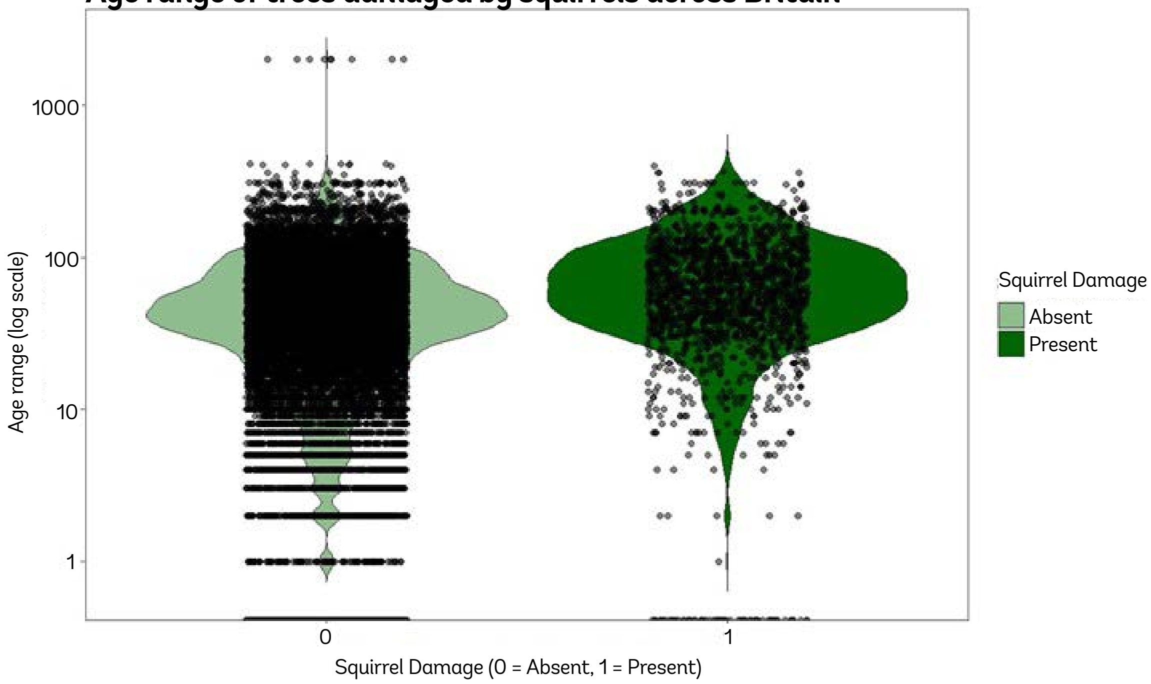

Figure 5. Violin and jittered density plot of age range of trees in woods damaged by squirrels across Britain.

The violin plot shows the whole distribution of the age range data. Each data point is the age range of trees within one NFI forest square. The widest section of the violin plot indicates the highest concentration of points at that age range. This illustrates that for NFI forest squares damaged by squirrels, a higher proportion of squares have a higher age range compared to squares not damaged by squirrels.

Grey squirrel impacts on UK woods and trees and woodland resilience

Our review has revealed substantial evidence gaps, indicating a lack of research into the impact of grey squirrels on UK woods and trees. The exception to this is the relative abundance of evidence on the negative economic impacts of grey squirrel damage on UK forestry. Evidence is particularly lacking regarding the impact of grey squirrel damage on the ability of UK woods and trees to deliver outcomes for biodiversity, air quality, water quality, flood management, recreation, access and wellbeing. Therefore, it is largely unknown whether grey squirrels have a net positive or negative impact on these outcomes. There is some evidence to underpin our understanding of the impact of grey squirrels on the ability of UK woods and trees to deliver outcomes for landscape character and tree species choice, and climate change mitigation and adaptation. However, key questions remain, including the magnitude of the impact of damage on these outcomes. The evidence, or lack thereof, is described further below, and is summarised in Table 1.

Economic

The most well-known impact of grey squirrels is the economic and physical impacts of bark stripping on timber quality. Several estimates of the economic cost of squirrel damage on the forestry industry have been made (Williams et al., 2010; Mayle et al., 2013). At a UK level, the estimated direct cost of grey squirrels is £40.6 million (Eschen et al., 2023). The majority of these costs were attributed to England (£32 million), compared with Scotland (£1.1 million, Wales (£1.1 million) and Northern Ireland (£2.5 million) (Eschen et al., 2023). This is compared with modelling that indicates a probable economic loss to forestry of approximately £37 million a year in England and Wales (RFS 2021; Richardson et al., 2021). This includes loss of timber value, reduced payments for carbon capture, the cost of mitigating damage (i.e. grey squirrel control), and the cost of replacing trees that have been fatally damaged. This is based on a range of assumptions, each of which impacts the real-terms cost of grey squirrel damage for woodland owners. The model takes squirrel damage frequency, severity and prevalence data from the NFI; and carbon capture loss estimates are acknowledged as being an overestimate (Richardson et al., 2021). Other costs such as squirrel control and replacing dead trees will vary depending on the management objectives of the woodland. Some managers will have a higher tolerance for tree mortality, especially if the primary objective is not timber production. Nevertheless, grey squirrels pose a huge economic impact on forestry and woodland management, undermining the confidence of woodland creation projects in meeting their objectives.

Wildlife

Since their introduction to Britain in the late 19th century, grey squirrels have negatively impacted the British landscape and wildlife. Grey squirrels extirpated and replaced populations of the sympatric Eurasian red squirrel *Sciurus vulgaris* (hereafter 'red squirrel') in much of Britain (Bertolino, 2008). As a result of competition with grey squirrels for resource, which is exacerbated and mediated by disease dynamics, red squirrels are now reduced to fragmented populations within Wales and England, with strongholds in Scotland and Ireland (Gurnell and Pepper, 1993, Gurnell et al., 2004, Rushton et al., 2006). Few studies have demonstrated negative consequences of grey squirrel presence on bird species in the UK (Hewson and Fuller, 2003), however the magnitude and distribution of these impacts is unknown and is thought to be insignificant (Newson et al., 2010; Broughton, 2020). Beyond the red squirrel, the impact of grey squirrels on other wildlife and biodiversity in the UK are not well studied.

Woodland ecological condition

The value of UK woodlands for wildlife and biodiversity is often measured in terms of ecological condition, which is evaluated based on a range of criteria. Whilst we do not know the direct impact of bark stripping damage on biodiversity, impact can be inferred from the likely effect damage has on these woodland ecological condition indicators (Forestry Commission, 2020).

Woodland ecological condition indicators on which grey squirrels are likely to have a NEGATIVE impact:

- Age distribution of tree species: Younger trees are more likely to die from bark stripping damage (Peden et al., 2020) which is likely to reduce the diversity of age classes present in woodlands (Mountford 2006). Also, woodlands with a higher age range of trees present are more likely to show signs of damage – see Figure 5.

- Herbivore damage: The presence of damage from herbivores, including grey squirrels, is recognised as having a negative impact on the condition of woodlands for wildlife (Forestry Commission, 2020).
- Occupancy and number of native tree species: Due to damage preferentially occurring on certain broadleaf species (Rowe and Gill, 1985; Peden et al., 2020), many of which are native, this disincentivises landowners from planting native broadleaf trees (Hemery et al., 2020), resulting in lower native tree species richness and therefore negative impacts on biodiversity.
- Tree health: Wounds created by squirrels can allow bacterial and fungal infections to enter (Abbott et al., 1977; Gill, 1992c; Mountford, 1997).

Indicators on which grey squirrels have the potential to have a POSITIVE impact:

- Open space: where squirrel damage results in the death of trees, this has the potential to produce open space within woodlands. However, the extent to which this is a positive impact is highly context dependent and won't necessarily lead to an optimal level of open space within woodlands for wildlife and plants.
- Vertical structure: the impact of bark stripping on the growth rate and the form of damaged trees (Gill, 1992b) means that damage is likely to result in a more varied woodland vertical structure.
- Deadwood volume: tree death from bark stripping can increase the volume of standing deadwood and loss of branches can increase the volume of deadwood on the woodland floor. The UK suffers from a dearth of deadwood in our woodlands (NFI 2020, Smart et al., 2024). However, the benefits of deadwood for biodiversity are most acutely delivered where there is a diversity of types, ages and species of deadwood, which is less likely to be optimal due to bark stripping. Also, the presence of grey squirrels is not a limiting factor in deadwood formation in old growth forests in North America (Gurnell et al., 2016).

Indicators where it is unclear if grey squirrels are likely to have a negative or positive impact:

- Veteran trees: the benefits to wildlife of veteran and ancient trees are largely due to the diversity and complexity of microhabitats and niches that old trees provide, in the form of hollows, cracks and apparent deformities such as missing branches or bark. Signs of bark stripping damage can often be the most obvious microhabitats available for invertebrates, bats, birds and fungi to utilise, especially in younger woodlands. In this sense, squirrel damage can act as a form of veteranisation of younger trees (Bengtsson, 2015). However, whilst damaged trees may display veteran features, they are unlikely to reach their full potential in terms of age and therefore will not become as valuable as true ancient trees.
- Woodland area: while grey squirrels can cause the death of individual trees,

they do not reduce the size of the woodland parcel itself.

- Invasive plant species: there is no evidence to suggest that grey squirrel bark stripping damage influences the presence or cover of invasive, non-native plant species.
- Proportion of favourable land cover around woodland: squirrel damage does not have an impact on land outside the woodland's boundaries.
- Woodland regeneration: the death of trees from squirrel damage may result in canopy gaps into which saplings can regenerate and squirrels distribute seed through caching behaviour. However, it is not clear if this has an overall positive impact on regeneration (Gill et al., 1995) because of the combined negative effects of death of young trees, seed predation and the habit of gnawing the seed's radicle before burying, which is the embryonic root of the seed, and therefore prevents germination and seed growth (Pigott et al., 1991).
- Vegetation (field and ground flora): there is no evidence to suggest that grey squirrel bark stripping damage influences the National Vegetation Classification of woodlands. While canopy gaps from pests/diseases have been shown to have an influence on ground flora species richness (Smart et al., 2024), the extent to which this is positive or negative depends on the context of the woodland.
- The impact of bark stripping damage likely has both positive and negative impacts on woodland ecological condition. These impacts may vary depending on the frequency and severity of the behaviour. Younger woodlands are likely to see short-term positive impacts due to creation of standing deadwood and increasing vertical woodland structure, however these impacts are unlikely to contribute to the overall resilience of the woodland over time. It is important to note that the impact of grey squirrels on woodland ecological condition (either negative or positive) is a significant evidence gap limiting our understanding of the ecological impacts of this invasive species.

Climate change mitigation

Woods and trees are important for sequestering carbon and contributing to climate change mitigation. There have been no studies directly looking at the impact of grey squirrel damage on the ability of woods and trees to sequester carbon dioxide from the atmosphere.

Damage from grey squirrels directly impacts the growth and form of a tree, leading to structural deformities and reduced crown size (Gill, 1992b), negatively impacting its ability to sequester and store carbon. However, tree growth rates can recover from less severe bark stripping incidents (Gill, 1992). Approximately 50% of a tree's dry weight is carbon, representing a significant store. For a tree to maximise its individual ability to store carbon, it must grow to its full potential. Young trees (<20 years old) are more likely to die as a result of damage (Peden et al., 2020) which means the residency time of carbon stored in these young trees will be reduced. Trees of 10-40 years exhibit the highest increase in rates of carbon sequestration throughout the lifetime of trees as measured by the Woodland Carbon Code (WCC, 2021) which is the age most frequently cited as vulnerable for damage (Middleton, 1931, Shorten, 1957, Rowe, 1967, Rowe, 1984, Rowe and Gill, 1985, Gurnell and Pepper, 1988, Gurnell, 1989, Kenward and Dutton, 1996, Rayden and Savill, 2004).

The timber value of a tree damaged by grey squirrels is greatly reduced. Damaged trees are likely to end up as woodchip or firewood, rather than being used in construction (Derbridge et al., 2016). This greatly effects the speed at which the carbon stored in the trees is released back into the atmosphere as trees used in construction can potentially lock up their carbon for centuries (Forster et al., 2019).

The impact of grey squirrel damage on carbon at the scale of the woodland is very hard to predict because of the complexities of woodland carbon dynamics. Around 75% of carbon in woodlands is found in the soil (Forest Research, 2015). The extent to which the flux of carbon in soil is net positive or negative depends on a range of factors, including soil type, level of soil disturbance, litter and deadwood inputs, root exudates, microbial activity, microclimate, speed of nutrient cycling and soil aggregate formation. The interactions between the impact of grey squirrel damage on woodland soil and these factors is currently unknown.

Air quality

Trees and woods have a role to play in improving air quality both in our towns and cities, and in the wider countryside. Trees are often used in urban areas as a way of redirecting the flow and forming a barrier between people and pollutants, such as nitrogen oxides from cars. They also remove some particulate pollution from the air by catching the tiny particles on their leaf surfaces. The urban heat island effect is a phenomenon where temperatures in towns and cities are artificially raised due to man-made structures such as buildings and roads absorbing and re-emitting heat. Trees have been shown to combat this effect by shading and reflecting radiation from the sun. Grey squirrel bark stripping damage is more often considered a problem in rural areas where the impacts are felt due to economic losses of timber. However, substantial bark stripping does occur in urban and peri-urban areas (Merrick et al., 2016), and the associated impact on growth and form of the tree (Gill, 1992b) is likely to reduce the effectiveness of their role in improving urban air quality and tackling high temperatures. This impact will be most pronounced where bark stripping is severe, or the tree dies from the damage.

We lack evidence regarding the level of severity of damage in urban compared to rural areas, so it is not possible to predict the extent of the impact on urban air quality. In rural areas, trees are often used to buffer sites of high conservation value, for instance ancient woodlands, from sources of nitrogen agricultural pollution such as ammonia (Bealey et al., 2016). Their ability to perform this task may be reduced if severely damaged by grey squirrels or trees die. However, where damage is less severe, trees can still develop, albeit with altered form and increased forks, which will still offer some buffer from pollution.

The effectiveness of damaged trees as buffers from sources of agricultural pollution has not been empirically tested, nor the impact of pollution on bark stripping behaviour itself. High levels of nitrogen pollution in habitats such as woodlands can have a fertiliser effect, disrupting ecosystem dynamics (Bobbink et al., 2010). Under increased levels of nitrogen, plants may find themselves at a competitive advantage and grow more vigorously in response to the increased availability of nitrogen in the environment where it was previously lacking or at low levels. Bark stripping damage tends to be more severe on vigorously growing trees (Kenward and Parish, 1986). Therefore, it is possible that nitrogen pollution in woods increases the risk of squirrel damage.

Water quality and flood management

The role of trees for improving water quality and flood management depends in a large part on their roots. Tree roots take up water from the soil which travels up the tree and is lost via transpiration into the atmosphere. Roots also improve soil porosity, allowing water to soak into the soil, rather than flowing straight into rivers, taking topsoil with it. Above ground, trees use their considerable surface area to intercept rainfall. This both reduces the speed at which rain hits the ground, which is good for protecting soil from erosion, and reduces the volume of rainwater that reaches the ground as water evaporates from the tree's leaves and bark.

Studies have shown that semi-natural broadleaf woodland can reduce rainfall-generated flooding (Monger et al., 2022) and it has been suggested that diverse structures in woods are beneficial for slowing the flow of water. The impact of squirrels on woods may promote a diverse structure in young even-aged broadleaf woodlands but this is likely to be a short-lived effect.

Trees can help to improve water quality by shielding soil from heavy rain erosion, reducing runoff during rain events, and by lowering windspeed at the ground level to prevent soil being lost to wind. Trees that have been damaged by squirrels usually exhibit structural deformities (Gill, 1992b) such as a shorter stature and increased forking. This is unlikely to prevent them from having the same outcomes for water quality as undamaged trees. Unless grey squirrel damage is severe and kills the tree, it is unlikely to have a large negative impact on the ability of trees to provide benefits for water quality and flood management.

Recreation/access/wellbeing

Grey squirrels are a charismatic woodland mammal that many people enjoy watching, representing a tangible connection to nature, especially those living in urban environments (Gurnell et al., 2016). The current distribution and habits of our native red squirrels means that it is hard to imagine them filling this role if grey squirrels had never been introduced. However, many reports and stories from the early 20th century (Coates, 2023) suggest that the benefits people get from seeing squirrels in towns and cities are not restricted to greys. Red squirrels were a common sight all over the UK and were common frequenters of gardens around the biggest of our cities such as London. Today, there is still a strong desire for the public to see red squirrels, regardless of whether they live in areas with red or grey squirrels (Dunn ***et al.***, 2021).

The impact of grey squirrels on people's access to woodland, and therefore their ability to access recreation and wellbeing benefits, has not been formally investigated. The main impact is likely to be one of health and safety. Squirrel damage can cause trees along paths to become weakened, making them more likely to collapse or drop branches. This adds work to the woodland manager's workload and may lead to paths being closed for tree safety maintenance, stopping people from enjoying publicly accessible woodlands.

There are no studies looking at the impact of observing squirrel damage on trees on people's perception of woods and therefore their ability to affect health and wellbeing benefits of people when in woodlands. However, public perceptions to grey squirrels and the threats they pose have been investigated, with 44% of 186 respondents perceiving grey squirrels as a high ecological risk (Gozlan et al., 2013). This is likely due to the impact of grey squirrels on red squirrel populations, rather than their damage to trees. However, a Mammal Society survey found that 45% of people liked grey squirrels compared to 24% disliking them (see Harris et al., 2006). Indeed, most people, if shown evidence of squirrel bark stripping, would not be able to attribute the cause of the damage. Increased awareness and education of the impacts of grey squirrels on trees is needed.

Landscape character and species choice

Landscape character is the unique combination of attributes that make each landscape different. In landscapes dominated by trees, the mix of tree species present plays a role in determining that landscape's character. Many factors influence which tree species are present in a landscape. For instance: geology, soil type, altitude, microclimate, and choices made by land managers about which species to plant or retain. Impacts of other mammals such as deer and squirrels also have a strong impact on the species present in a landscape.

Individuals of species more susceptible to damage are at a competitive disadvantage. It has been suggested that this results in a change in canopy composition over time (Mountford et al., 1999), which could ultimately impact the landscape character of areas such as the beech-dominated chalk hills of the Chilterns, or the oak-dominated temperate rainforests of Wales and Devon. However, direct evidence of a change in canopy composition as a result of squirrel bark stripping is lacking. Indeed, the latest Bunce survey indicated that over the last 50 years the rank order of broadleaf canopy dominants has stayed remarkably stable, with oak remaining the most common broadleaf tree species found across Great Britain, and sycamore consistently in fifth place (Smart et al., 2024). Beech has remained relatively stable in occurrence in Britain's broadleaf woods over the last 50 years, despite dropping down the rank order due to an increase in holly (Smart et al., 2024). This, however, is due to the impact of climate change on holly occurrence, rather than the impact of grey squirrels on beech trees.

The threat of grey squirrel damage is a key factor found to be discouraging or preventing expansion of tree cover among woodland owners and agents (Hemery et al., 2020). This impacts tree species selection in new woodland creation schemes and represents a shift away from planting vulnerable native broadleaf species (Huxley, 2003). This is likely to have a much bigger impact on the composition of our woodlands than altered natural processes, such as competition, as a result of grey squirrel damage. Native broadleaf species have high biodiversity value (Mitchell et al., 2019). If woodland managers are deterred from planting these species there will be a knock-on impact on the biodiversity value of newly created woodlands due to changing species composition.

Climate change adaptation

A changing climate is already influencing the distribution and occurrence of tree species in Britain (Smart et al., 2024). The impacts of rising temperatures and climatic changes will affect the range of tree species differently, with many predicted to shift their distributions to higher latitudes (Chen et al., 2011). Most native broadleaf tree species are at the northern edge of their range which means local extinctions are not predicted (Ennos et al., 2019). However, changes in growth rates, phenology and other functional traits are possible. A key factor promoting resilience of woodlands to these changes is natural regeneration, where trees are encouraged to self-seed, utilising their innate genetic diversity to track the impacts of changing climatic conditions (Cavers and Cottrell, 2014). The risk of premature death of trees before reproductive maturity posed by grey squirrels, may prevent their ability to set seed, and reduce the potential gene pool for populations to adapt to climate change.

Where damage is not severe enough to kill trees, it remains to be seen how the impact of climate change interacts with the associated threats faced by our woodlands. There have not been any studies investigating the impacts of grey squirrel damage on the ability of woodlands to adapt to climate change. However, it is clear that several threats posed to woodlands as a result of the changing climate will likely be exacerbated by squirrel bark stripping activity, such as:

- Drought: when bark is stripped by squirrels the exposed tissue is at increased risk of desiccation, making damaged trees more likely to be susceptible to drought.
- Storm events: damaged trees are more vulnerable to wind-snap at the crown (Gill et al., 1995, Gill, 1992c; Mountford, 1997; Huxley, 2003)
- Pests and diseases: wounds created by squirrels can allow bacterial and fungal infections to enter (Abbott et al., 1977; Gill, 1992c; Mountford, 1997).

Table 1. Summary table of the main impacts of bark stripping on various ecosystem services.

Ability of trees and woods to deliver outcome	Positive or negative impact*	Scale of impact*	Strength of evidence	Key message
Economic		High	High	At a UK level, the estimated direct cost of grey squirrels is £40.6 million with a probable economic loss to the forestry industry of approx. £37m/year in England and Wales.
Wildlife and woodland ecological condition	?	Unknown	Low	Negative impacts on red squirrels well established, wider biodiversity impacts unknown. However, some ecological condition indicators will be impacted.
Climate change mitigation		Low	Medium	Damaged trees are likely to sequester less carbon throughout their lifespan. The impact on overall woodland and soil carbon flux is unclear.
Air quality	?	Low	Low	It is unknown if the impact of damage on the growth and form of the tree will impact its ability to deliver air quality benefits.
Water quality and flood management	?	Low	Low	Unless severe damage occurs, it is unlikely to have a negative impact on the benefits of trees for water management, but evidence is lacking.
Recreation, access and wellbeing	?	Low	Low	Attitudes to grey squirrels are mixed and the impact of damage on woodland accessibility due to health and safety concerns is unknown.
Landscape character		High	Medium	It is thought selective damage of certain tree species will alter woodland composition and deter planting of broadleaf species.
Climate change adaptation		High	Medium	Squirrel damage is likely to exacerbate the impact of threats from climate change and may impact adaptation if regeneration is affected.

What is being done to tackle the problem of grey squirrel bark stripping damage?

At present, lethal control is the only available option to manage levels of squirrel damage in British woodlands. Lethal control is most effective if carried out collaboratively, at large, landscape scales (Thompson and Peace, 1962), requiring the cooperation of adjacent landowners. However, lethal control has also proved effective at managing the impacts of grey squirrel damage at the scale of the woodland if resources and funds are available to follow a rigorous squirrel management plan (RFS, 2021). Lethal measures include live and kill trapping, shooting and drey poking (Gill, 2019). Some success was found using warfarin to control grey squirrel populations (Mayle and Broome, 2013), however this is no longer approved for use in the UK due to welfare issues. Whilst lethal control is labour intensive, it can be effective in reducing levels of bark stripping damage if carried out at the right time of year and repeated annually within a woodland (Gill et al., 2019). Grants of £60 per hectare are available from Defra to help landowners manage grey squirrel impacts on their land.

Lethal control of wild animals is often controversial and generally not an accepted method by the public. Members of the British public found lethal control for squirrels largely unacceptable, however individuals within volunteer groups are often motivated by the conservation of red squirrels and are more accepting of control methods (Dunn et al., 2018, Dunn et al., 2021). For practitioners, volunteers and researchers involved with lethal control, there are often trade-offs between cost, effectiveness and humaneness of control methods (Crowley et al., 2018).

Historically, research into squirrel bark stripping behaviour has focused on monitoring the impacts of bark stripping, investigating control and prevention, and understanding the underlying causes of bark stripping and factors that could be used to predict the behaviour (Nichols and Gill, 2016). Currently, the main research efforts to tackle the problem include the following, all of which have the ultimate aim of reducing the impact of bark stripping damage on UK woods and trees:

- Investigations into chemical attributes of trees that may influence the likelihood of different tree species being damaged: Whilst not a new concept (Kenward and Parish, 1986), recent studies have indicated promising results (Ash, A. Pers. Comm, 2024). The identification of chemical compounds that deter squirrels (either by mechanisms of taste or smell) from bark stripping provides new opportunities to prevent squirrels from damaging trees. Chemicals likely to deter squirrels may inform methods to protect trees through the physical application of identified compounds to growing trees (such as chemical sprays or paints) or long-term gene-editing methods that could exploit the identified resistant traits in future tree breeding programmes.
- Fertility control to prevent grey squirrel reproduction: The UK Squirrel accord is currently championing and supporting a project led by the Animal and Plant Health Agency to develop an oral contraceptive. The aim is to reduce population size and the rate of population recovery after culling, for which fertility control has shown potential (Massei and Cowan 2014), especially in combination with initial short-term culling (Croft et al., 2021).

Challenges to development include: a species-specific delivery mechanism, designing a contraceptive that is effective orally, validation testing on grey squirrels which are known to be difficult to breed in captivity, determining the optimal density of feeding hoppers to deliver the contraceptive, and the most effective bait to attract enough of the population to have an impact (UKSA, 2024). The UKSA anticipates the fertility control to be available for use in the field by 2030 (UKSA, 2024).

- Gene drive technologies as a novel method of population control: Gene drive skews the inheritance ratio of an allele so that it spreads quickly through a population (Burt, 2003). In the case of population control, this means ensuring all offspring are male so that the number of breeding females eventually dies out and modelling has shown its potential effectiveness (Faber et al., 2021). This is known as directed inheritance gender bias and is being championed and supported by the European Squirrel Initiative (ESI, 2021). Gene drive technology may offer a humane, efficient and cost-effective method of control. It has the benefit of being relatively cheaper to develop than fertility control and may be more likely to result in local extinctions (Whitelaw, 2021). However, development of a working product is not as far along as research into fertility control.
- The impact of native predator recovery in the landscape such as pine marten on squirrel numbers: evidence is emerging that recovery of pine marten populations is resulting in declines in grey squirrel numbers to the benefit of red squirrels (Sheehy & Lawton, 2014; Sheehy et al., 2018). Pine martens evolved alongside red squirrels, whereas they represent a novel predator for grey squirrels. This has been indicated by response to pine marten scent, which red squirrels avoid, and grey squirrels do not show any anti-predator behaviours (Twining et al., 2020a). Pine martens won't necessarily be the answer in all environments; for example urban areas will likely act as strongholds for grey squirrels (Twining et al., 2020b). However, in broadleaf woodlands healthy pine marten populations allow red squirrels to thrive due to grey squirrel declines (Twining et al., 2022) with the concomitant benefit of reduced bark stripping damage. It has been suggested that goshawks may also have a similar impact on grey squirrel numbers, but more evidence is needed to test this at a population scale.
- The impact of grey squirrels on wider ecosystem services: improved understanding of the wider impacts of grey squirrels on nature and society will help mitigation of these effects and help to bolster the case for grey squirrel management to policy makers and the public. A PhD is currently in progress investigating the impact of grey squirrels on ecosystem services, led by the University of York and the Forestry Commission.

Conclusion

In a relatively short space of time, grey squirrels have become a major threat to the resilience of broadleaf woodlands in the UK. While impacts of damage beyond economic cost (timber losses) are largely under-studied, it is likely that damage will interact with and exacerbate the impacts of climate change on woodlands and negatively impact their ability to adapt. Bark stripping damage is also likely to have a significant impact on the effectiveness of trees in mitigating climate change, and potential negative impacts on woodland biodiversity. Currently, novel research is investigating how to effectively reduce grey squirrel numbers and understand mechanisms underpinning bark stripping behaviour. Novel approaches to investigating grey squirrel behaviour, ecology and impacts within UK woodlands will facilitate the development of effective and targeted management methods to improve woodland resilience and adaption to climate change.

Key evidence gaps

Much previous research has focused on the key tree-level factors of susceptibility such as age, size and species. Because the impacts are thought to be extensive, research has concentrated on methods to mitigate and reduce damage rather than understanding the drivers of the behaviour.

Although we assume the impacts of bark stripping are largely negative (which they likely are), we do not understand the scale or complexities of these impacts.

Currently, we have no information of bark stripping impacts on:

- Biodiversity and nature recovery
- Ecological condition of woodlands
- Climate change mitigation
- Air quality
- Water quality and flood management
- Recreation/access/wellbeing
- Landscape character and species choice
- Climate change adaptation.

Further, it has been widely discussed in the literature that grey squirrel population density likely has direct influences on the frequency and severity of bark stripping. However, there is no detailed or fine-scale data on grey squirrel population density across the UK. Regional-level data for population density would be immensely powerful in assessing the relationship between density and damage and could provide useful recommendations for management and potential information on drivers of the behaviour. Large-scale data on squirrel damage and population density alongside climate and tree mast data could help produce tools to predict when damage may be likely to occur or increase, allowing woodland and forest managers to design appropriate management plans.

Additionally, the underlying causes of bark stripping behaviour are not known. The difficulties of studying wild animal behaviour have prevented progress in understanding squirrel behaviour. Squirrels are dynamic, arboreal species that spend most of their time in the canopy of forests. Squirrels can be ubiquitous in the settings of parks and gardens, however in remote country forests they are often elusive and difficult to observe. More direct observations of bark stripping behaviour in the wild are needed to unravel the complex nature of what drives squirrels to bark strip. Knowing why squirrels bark strip, how often and which individuals are performing the behaviour is crucial in developing effective management.

Lastly, many methods of grey squirrel management are being developed in the UK currently. These range from predator introductions, oral contraceptives and gene editing techniques. The long-term sustainability of these methods, which are largely humane methods to manage wild squirrel populations, is unclear. These methods are favoured among members of the public and utilise exciting novel technology and research. Additionally, the topic of squirrel welfare has largely been absent in public and academic debates. Care needs to be taken with the welfare impacts of any future methods on grey squirrels, which still remains a large evidence gap. Yet, the effectiveness of these newly developing methods in reducing the frequency and severity of bark stripping remains to be proved.

What needs to happen

Over the past couple of decades, research into bark stripping behaviour has slowed, although, in the last five or so years, it has started to pick up again. With squirrel damage identified as the 'number one threat' to broadleaf woodland health by the RFS, it is clear that funding should be directed to ameliorate the above evidence gaps. Woodland creation grants are incentivising woodland expansion, adding a renewed sense of urgency to tackle the grey squirrel problem to enable the maximum benefits of these new woodlands to be realised. As squirrel control continues to be uncoordinated at the landscape scale, decision-support tools made for landowners would be helping in assessing risk of squirrel damage to their crop and how to target management effort. While the impacts of bark stripping are well understood in the forestry community, a vast majority of the members of public are unaware of the detrimental impacts that squirrels can have on our native broadleaf trees and forests. More education of the problems caused by invasive grey squirrels would be beneficial in raising awareness of their impacts and gaining support from the public in finding a solution to squirrel damage.

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Extreme weather events

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Abstract

With the changing climate we are expecting to see an increased frequency and intensity of extreme weather events including fires, storms, drought and floods. It is thought that these events will have impacts on our trees and woods (both positive and negative), and we will need to plan to adapt to make them more resilient to these perturbations.

Overall findings:

- Climate models show a high probability of increased frequency and intensity of extreme weather events.
- No long-term datasets on effects of extreme weather exist to show if these events are currently increasing with climate change.
- There is very little data available on the impacts of these extreme events on our native woods and trees. Most evidence comes from commercial forestry and may not be transferable to native wooded habitats.
- Woody habitats are thought to be sensitive to the effects of climate change and extreme weather, especially if degraded.
- Some data is available on the economic, health and other social impacts of these events, but more is needed to help us understand impact and plan mitigation measures.
- Although on balance the impacts are likely to be negative, how we consider plausible positive impacts in the future, and how this relates to adaptation planning, needs further work.
- We need to clearly identify evidence gaps and fill them to help inform the development of adaptation plans.

Introduction

The UK Climate Projections 2018 (Met Office, 2018) show that the projected climate change trends over land for the 21st century are for increased chances of warmer, wetter winters and hotter, drier summers, as well as increases in the frequency and intensity of extremes.

We hear more and more frequently about these extremes in the news:

Record-breaking weather extremes in 2023/early 2024

- Global temperatures breached 1.5°C for a record 12-month period (at 1.52°C).
- February 2024 was the hottest month in human history – a full 1.77°C above pre-industrial temperatures.
- Worst ever wildfires in Canada, Hawaii, Europe.
- Record temperatures in Arizona and South West USA, France, Germany, Poland, Australia and so on.
- Heatwaves in the middle of winter in Chile and Argentina.

- Highest rainfall levels (and consequential flooding) in many countries.
- Levels of sea ice in Antarctica at a new low in both the summer and the winter.
- Global average ocean surface temperatures "off the charts" – as warm in February 2024 as would typically be the case in mid-July.
- A record number of billion-dollar climate-induced disasters in the USA in 2023.
- Hottest May, June, July, August, September, October, November, December 2023, and January and February 2024. A total of 140 countries broke February heat records.

Understanding the impacts of these extremes on our trees and woods is vital to allow us to plan and adapt and increase resilience to these perturbations.

Methods

A non-exhaustive literature search was conducted for extreme weather events using Web of Science and Google Scholar to find both published peer-reviewed literature and grey literature.

Search string used was: 'forest*' OR 'wood*' OR 'tree*' AND 'weather' OR 'flood*' OR 'storm' OR 'drought' OR 'wind' OR 'fire'.

Reference lists were checked to see if they contained any additional relevant studies.

Studies from outside the UK were included if they fall into similar climatic and ecological conditions; these studies are noted in the review.

The conservation evidence database and Applied Ecology Resources were also checked for relevant literature, as were several organisations' websites including Forest Research, Met Office, Natural England, Natural Resources Wales and Nature Scot.

Photos and figures of damage on the Woodland Trust estate were gathered from colleagues.

Results

The UK Climate Projections 2018 (Met Office, 2018) show that the projected climate change trends over land for the 21st century show increased chance of warmer, wetter winters and hotter, drier summers, as well as increases in the frequency and intensity of extreme weather events. Extreme weather events include wildfire, drought, floods and storms. The UK Met Office report on climate extremes (UK Met Office, 2018) shows that already the average length of a warm spell is increasing from 5.3 days (1961–1990) to 13.2 days (2008– 2017), with rain falling by 4% - 17% over the same period.

A recent report from Natural England (Staddon et al., 2023) used expert assessment to understand which habitats are likely to be the most sensitive to the impacts of climate change such as extreme weather events (the sensitivity of a habitat to climate change is defined as: *the outcome of the inherent sensitivity and adaptive capacity of the habitat to environmental changes*). Habitats were ranked on a five point scale, with five being the most sensitive to change. Almost all habitats were ranked as medium to highly sensitive, indicating changes to these habitats are likely to occur under current climate projections. Habitats that were degraded (i.e. in poor ecological condition) were considered more sensitive to the impacts of climate change. Wooded habitats that were ranked as high sensitivity in both a good and degraded condition include lowland beech and yew woodlands, wet woodlands and native pine woodlands. Some habitats had a large difference in sensitivity scores between degraded and good condition, including some woodlands that scored between two and three in good condition but four when degraded. This highlights that degradation has an important influence on sensitivity.

In terms of extreme weather impacts, the following types of wooded habitats were thought to have extreme weather-related risks:

- Wood pasture and parkland: resilient, but risks from storms
- Upland oakwood: drought, wildfires
- Lowland beech and yew woodland: drought, wildfires
- Upland mixed ash wood: storms, drought
- Wet woodland: drought, wildfires
- Lowland mixed deciduous woodland: good resilience, but risks from drought, wildfires
- Upland birchwoods: drought, warming
- Native pine woodlands: drought, wildfires, storms

Fire

Wildfires are defined in the UK as 'any uncontrolled vegetation fire which requires a decision, or action, regarding suppression'. In recent years several large-scale UK wildfire events have led to heightened interest in building an understanding of their behaviour and impacts.

Wildfire frequency and intensity can be driven by vegetation type and weather patterns and can also increase with disease outbreaks, drought, windthrow damage and hotter and drier conditions. The latest UK Climate Projections (UKCP18) shows that the risk could double with a 2°C global temperature increase and quadruple under a 4°C temperature increase (UK Climate Change Risk Assessment (CCRA3) Evidence Report 2021).

Wildfire risks are also linked to changing land management practices and human behaviour. In the UK today, wildfires are considered a semi-natural hazard as most wildfires in the UK are started by human activity (Woodland Trust Estate data 2024, Gazzard et al. 2016).

Globally, wildfires have been highlighted as one of the major disturbances that negatively impact ecosystem services in terrestrial habitats, including forests and woodlands (Thom and Seidl 2016). These impacts include:

- damage to or loss of habitats and species
- air pollution which has human health impacts (HECC report 2023)
- trigger flooding and landslides
- water pollution
- soil erosion (Shakesby 2011), runoff (Vieira et al., 2015), water quality (Harper et al., 2018), and soil fertility (Caon et al., 2014)

- long-term damage to soils from fire can release significant amounts of carbon and greenhouse gases into the atmosphere (UK Climate Change Risk Assessment (CCRA3) Evidence Report 2021; Belcher 2021)

The Climate Change Risk Assessment (CCRA) has defined wildfire risk as 'cross-cutting', meaning it has implications for the natural environment, people, buildings, infrastructure and businesses, and crosses the rural-urban interface (CCRA 2021).

Whilst wildfire is known to have these negative impacts, it can also have some positive impacts that enhance ecosystem services such as regeneration of vegetation and control of pests and diseases (Pausas and Keeley 2019). Because of this, humans have used controlled burning regimes for management across the UK for centuries such as in heathlands and moorlands.

Frequency and impact of wildfires in the UK

The reporting that wildfires are increasing with climate change is often still anecdotal (for example, senior officials in the UK Fire & Rescue Services report there is a longer wildfire season, and an increasing spatial scale of wildfires) but there are no consistent, long-term datasets in the UK that allow us to link wildfires to climate change (Belcher et al., 2021).

Satellite data for the years leading up to 2020 saw an apparent increase in the number and area of burning incidents. This data appears to show some correlation for wildfire occurrence with mild winters, higher temperatures, heatwaves and prolonged dry periods across spring and summer. Periods with low wildfire incidences correlate with heavy periods of rainfall in spring and summer as well as wetter winters (Woodland Natural Capital Accounts 2020).

Forestry Commission England began publishing wildfire statistics in 2009. Between 2009-2021, Fire and Rescue Services attended over 360,000 wildfires in England, with an average of over 30,000 incidents per year. Over 79,000 hectares of land was burnt, and the wildfires burned for just under 540,000 hours. Over this period, the majority of wildfires in woodland were in broadleaved woodland (Forestry Commission 2021). Between 2009-10 and 2016-17 woodland and forest fires accounted for less than 5% of the land area burnt in England. The vast majority of wildfires in the UK occur in areas with low shrub vegetation (i.e. lowland and upland heath).

What is causing this increase?

It has been suggested that changes to wildfire activity in the UK may be explained by a change to fuel, weather and ignition conditions. Climate change will influence all three of these factors. The predicted warmer, wetter winters will result in a longer growing season and therefore a greater abundance of vegetation. On top of this, more frequent and longer-lasting heatwaves will lead to drier vegetation, increasing the amount of fuel available to burn. It is also thought that warmer summers will lead to more people participating in activities such as barbecues, a major source of ignitions. A study modelling climate change impacts for the Peak District found that a combination of higher temperatures and a rise in the risk of ignition due to increased recreation will likely lead to more summer wildfires towards the end of this century (Albertson, 2010).

Forestry guidance in the UK has been working towards providing wildfire mitigation advice for at least a decade. The aim is to prevent the build-up of fuel across entire individual sites or at strategic locations identified by wildfire management planning (Forestry Commission, 2014).

This can include:

- The use of timber thinning and harvesting to remove timber and disturb surface fuels to improve resilience.
- Environmental management to remove heather, gorse and other materials, and cutting of grass or corn during cropping and harvest.
- Using different planting regimes, planting belts of less fire-prone vegetation around more flammable units.
- Utilising firebreaks and defensible zones around infrastructure to provide wildfire adaptation and prevent large fire spread.

Drought

The projected warmer temperatures and drier summers are expected to lead to an increase in the frequency of drought and heatwave events in the UK, with the largest risk being in southern England. Risk of drought is influenced by:

- Timing – dry springs may affect the current year's growth and late summer droughts may affect the next season's growth more, with repeated droughts having a cumulative effect, leading to growth reductions several years later.
- Location – summers are projected to become drier across most of the UK, with the largest reductions in southern England, where temperatures will also be warmest.
- Site soil – shallow, light-textured and freely draining soils will hold less water than deep, heavier soils, and therefore trees growing on shallow, lighter soils are more prone to water stress and drought impacts.
- Tree species – drought sensitivity varies between tree species. Species such as beech, birch and sycamore are more sensitive than hornbeam or native oak species. Differences in drought tolerance can also exist between different local populations.
- Root depth – ground vegetation and shallower rooting shrubs and trees may be more affected than deeper rooted tree species. Newly planted trees will be particularly vulnerable to droughts, especially those on exposed open sites and smaller, more fragmented woodlands are more likely to dry out than extensive areas.
- Age – newly planted, older and veteran trees are more likely to be affected by drought.
- Condition – forest recovery after a drought disturbance is linked to the condition before the disturbance.

Drought is known to increase water stress in trees and can lead to reduced growth, crown dieback and even death. There are also knock-on impacts such as an increased risk of windthrow, wildfire, and pest and disease outbreaks, as well as reduced carbon sequestration due to reduced growth.

The risk of drought needs to be assessed and adaptation measures put in place to help manage these risks. Research and ongoing monitoring can help better understand the risk and impacts of drought on different tree species, provenances and sites. Drought may have consequences for the location for tree planting as there has been emerging evidence to suggest that trees may compete for water with agriculture and human consumption in dry, lowland regions (Tew 2019).

Forest Research is currently undertaking drought risk research including through the PRAFOR (Probabilistic drought Risk Analysis for FORested landscapes) and FORWaRD (Forestry and Woodland Resilience to Drought) projects. The FORWaRD project will provide vital information regarding the following questions:

- How does species drought tolerance vary, and can we match this with site drought risk for robust afforestation decisions?
- How resistant and resilient to historical extreme drought events are UK tree species, can different species mixtures mitigate the impacts of these events, and how does ecosystem complexity influence this resilience?
- How do ancient semi-natural woodlands and those recently established on ex-industrial or ex-agricultural land differentially respond to a range of past extreme drought events, and how quickly do they recover following disturbance?
- What are the likely impacts of future climate shifts on species response to drought, and how best can we map the risk?

Storms and windthrow

Storm damage has economic, social and ecological impacts including tree growth, safety concerns and restriction on public access. Forest Research assessed the impact of storms in Great Britian in 2022 and showed almost 12,750 hectares of tree loss, with approximately 3,350 hectares of damage recorded in England. The majority of the damage was as a result of Storm Arwen in November 2021. The overall damage was found to be relatively modest, equating to around 0.2% of England's tree cover. FR said that over 90% of trees that are lost in storms will be replanted, meaning only a small percentage of forest is actually lost in the long term where it is not possible to restock. At a national scale, the level of loss is comparatively modest, but the loss of trees can also have a devastating impact on individual woodland owners.

Wind is a major threat to woodlands and forests across the United Kingdom, particularly in upland and western areas. Climate change projections include an increased frequency of storms, an increase in wind speeds and increased rainfall, which are expected to increase the risk of windthrow (Met Office, 2018; UK Climate Change Risk Assessment (CCRA3) Evidence Report 2021).

Windthrow risk is influenced by:

- The exposure of the site, with higher wind speeds occurring in western areas and at higher elevations.
- Tree species and rooting form, with both shallow rooting and faster growing species more vulnerable.

- Soil type, drainage, and rooting depth; with shallow, water-logged, or sandy soils at increased risk.
- Stand spacing; open grown trees exposed to wind are at a lower risk, as are closely spaced (<2m) stands.
- Tree height.

ForestGALES project at Forest Research helps with predictions of windthrow risk, but only for commercial conifer forests; there appears to be no equivalent for native wooded habitats.

When people think of storms in Britain, they often think of October 1987 when a storm was experienced with wind speeds thought only to be likely every 200 years and locally gusting to 160kph. An estimated 15 million trees were blown down across South East England. Following the event, a study of woodland sites that were exposed to the storm analysed ecological changes between 1971 and 2002, as well as ecologically equivalent sites that were not (Smart et al., 2014). Researchers found that although the impacts of the 1987 storm were spatially variable in terms of impacts on woody basal area, the storm had a positive effect on understorey species richness through the creation of gaps. The Bunce report (Snart et al., 2024) has shown that storm impacts are increasingly being recorded in long-term monitoring projects. This could be an important source of gap creation in woodlands, especially since management interventions for gap creation are rarer than they used to be.

Flooding

Many parts of the UK are already seriously impacted by flooding and the frequency and severity of floods is expected to increase with climate change due to a combination of projected sea level rise, heavier rainfall episodes and increased rainfall in winter.

Flooding and prolonged waterlogging can affect trees in many ways:

- Restricting the supply of oxygen to tree roots, preventing their normal function in absorbing water and nutrients.
- Restricting rooting depth and increasing the risk of windthrow due to reduced tree stability and anchorage.
- Damaging or washing away newly planted trees, before their roots become established.
- Reducing tree growth.
- Damaging soil health.
- Increasing vulnerability of trees to disease and infection.

Forests and riparian woodland can contribute substantially to reducing downstream flood risk due to their rainfall interception, water uptake and high surface roughness, slowing both runoff and the peak flow in streams and watercourses (see natural flood management chapter). However, these benefits are dependent on having woodlands and trees in good condition.

Discussion

Overall, the findings of this research show that an increased frequency and intensity of extreme weather events could potentially have a large impact on wildlife, pollution, ecosystem services and human health. This is based on climate models and knowledge of impacts on tree individuals as well as a few case studies, however, there is currently very little evidence on the impacts of extreme weather events on our native trees and woods. Further research and long term and consistent monitoring of extreme weather events would increase our understanding and help inform planning for mitigation, adaptation and response.

Evidence gaps

1. There is still a lack of long term and consistent data monitoring extreme weather events in the UK. This makes it difficult to understand if a changing climate is correlated with an increase in these events.
2. There is also a lack of data on the impacts of events on native woodlands. Knowledge is, and predictions of impacts are, based on what we know of impacts on stresses on trees or case studies of one or two sites after an event. Most research focuses on single events rather than whole regimes. We know how weather events may stress or damage trees but impacts on whole woodlands or landscapes are not thoroughly assessed.
3. Carbon emissions from burning woodland could impact UK greenhouse gas emissions. There is no evidence currently from the UK, but global studies have linked wildfires to loss of life, injury and respiratory distress, exacerbating chronic conditions and causing long-lasting psychological effects. The UK's high population density means that the risk could be high.
4. The UK does not routinely investigate the cause of wildfire ignitions and has no fire investigators qualified in this specialist skill. This is an area of concern as it casts significant doubt onto any of the current data sets that record an ignition source, making studies of social interactions as an ignition source impossible. As such more research is required in this area.
5. Assessment of what changes we might expect in our ecosystems in terms of vegetation changes because vegetation is the fuel for fire.

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Funding and skills gaps

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Introduction

The need to protect and restore biodiversity is a well-accepted idea. Protecting and restoring biodiversity can also yield economic benefit at both local and national level. Analyses have shown that half of global GDP is moderately or highly dependent on nature and its services (WEF, 2020; Dasgupta, 2021). However, a report by the United Nations Environment Programme (UNEP), (2023) estimated that the amount of money spent on activities that degrade nature is around $5 trillion per year, whilst nature-positive funding totalled around $200 billion a year.

In the UK, the decline in nature is predicted to result in a 12% loss of GDP over the coming years, more than the 2008 financial crisis and the Covid-19 pandemic (Green Finance Institute 2024). In addition, unmitigated climate change is predicted to cost the UK billions of pounds each year (Climate Change Committee 2021). Investing in the restoration of nature and combating climate change could help to mitigate these two threats as well as providing green jobs in the process of restoring nature- rich habitats.

One way of thinking about the economics and investment surrounding biodiversity and climate is natural capital. The UK Government's definition of natural capital is:

Natural capital includes certain stocks of the elements of nature that have value to society, such as forests, fisheries, rivers, biodiversity, land and minerals. Natural capital includes both the living and non-living aspects of ecosystems. Stocks of natural capital provide flows of environmental or 'ecosystem' services over time. These services, often in combination with other forms of capital (human, produced and social) produce a wide range of benefits. These include use values that involve interaction with the resource and which can have a market value (minerals, timber, freshwater) or non-market value (such as outdoor recreation, landscape amenity). They also include non-use values, such as the value people place on the existence of particular habitats or species.

In the UK, the natural capital of various habitats including woodlands, are calculated by the Office for National Statistics. The most recent woodland natural capital accounts calculated an economic value for the UK's trees and woods at £382 billion (Office for National Statistics 2024). This is split into various categories including:

- Greenhouse gases – UK woodlands sequestered 19.6 million tonnes of greenhouse gases, valued at £5.1 billion.
- Health and recreation – annual health benefits from recreation in woodland were estimated at £1.1 billion.
- Air quality – UK woodlands removed over 316,000 tonnes of air pollutants worth an estimated £1.8 billion in avoided negative health impacts.
- Flood regulating – the annual value of trees in reducing flooding risk in the UK was estimated at £911 million.

- Urban heat regulating – the annual value of city trees in providing shade was estimated at £753 million.

Some benefits cannot have a monetary value assigned to them and this is important to bear in mind when talking about nature in a purely economic way.

As well as having a good understanding of the economic value of the UK's woods and trees, reports have also shown that there is a positive cost-benefit relationship for restoring woodlands (RSPB, 2021). Specifically for woodland the report found that:

- Every £1 invested in afforestation is expected to generate £2.79 of economic and social benefits (through carbon sequestration, recreation, air pollution removal, timber and biofuel production and biodiversity support).
- 25 temporary jobs are expected to be created for every 100ha of the tree planting stage.
- Ongoing maintenance of woodland is expected to generate £314,000 in gross value added (GVA) per 100ha of planted woodland, over 100 years.
- Non-monetised benefits include enhanced water quality, noise mitigation, temperature regulation, reduced flood risk and improved biodiversity.

Investment in nature is essential. If we hope to mitigate all the direct and indirect threats our woods and trees face and meet legally binding nature and climate targets, there is a need for adequate resourcing through investment directly into nature recovery in a long-term skills base.

In the UK, a major way in which funding for this work is provided is through government grant schemes. Grant schemes and rates will affect what can be created, managed and restored. These grants therefore have a major influence on the future of our woodlands, their condition and resilience to future change, and their ability to continue providing ecosystem services.

Given the scale of investment needed, in addition to government schemes, private finance is increasingly taking a more prominent role. The Global Biodiversity Framework refers to the need to substantially increase funding from all sources by multiple mechanisms including:

- Leveraging private finance, promoting blended finance, implementing strategies for raising new and additional resources, and encouraging the private sector to invest in biodiversity; including through impact funds and other investment and;
- Stimulating innovative schemes such as payment for ecosystem services, green bonds, biodiversity offsets and credits, benefit-sharing mechanisms, and environmental and social safeguards.

Private finance may be better suited towards issues that have clear monetary value such as carbon or flood protection, with public and charitable funding better suited for nature restoration that doesn't have an easily attributed monetary value, and which require concerted action.

In addition to having the money to aid nature protection and restoration, there is also the need for a workforce capable of undertaking the extensive and sensitive work required to meet the legal commitments to reverse nature's decline by 2030 and meet net zero by 2050. Society also increasingly needs sources of skilled, long-term employment. Investing in green jobs is a win-win situation for community, economy and nature.

In this section we will explore the current funding available for woodland restoration, creation, protection and management from both the public and private sector. We also look at the current analyses of skills within the sector to try and understand whether there is the capacity to carry out sensitive and quality restoration and creation projects.

Methods

Grant information was collated during the period January to August 2024 by consulting the websites of the following grant-awarding bodies:

- The Tree Council Grants and Guidance section https://treecouncil.org.uk/grants-and-guidance/ (2024)
- The Woodland Trust Trees for Landowners and Farmers MOREwoods frequently asked questions - Woodland Trust (2004)
- UK Government website for information on Forestry Commission grants, sustainable farming incentive (SFI), countryside stewardship and landscape recovery fund funding for land or farms - GOV.UK (www.gov.uk) (2004)
- Scottish Government forestry grant schemes Scottish Forestry - Forestry Grant Scheme (2024)
- Scottish Government rural payments and services Agri-Environment Climate Scheme (ruralpayments.org) (2024)
- DAERA forestry grants DAERA Forestry Grants | Department of Agriculture, Environment and Rural Affairs (daera-ni.gov.uk) (2024)
- DAERA environmental farming scheme Environmental Farming Scheme (EFS) | Department of Agriculture, Environment and Rural Affairs (daera-ni.gov.uk) (2024)
- Welsh Government rural grants and payments Rural grants and payments | Sub-topic | GOV.WALES (2024)

The information collated was correct as of August 2024. However, it is important to note that all UK governments are currently revising their forestry and environmental schemes and developing new farming and land management subsidies following Brexit. All four countries are currently in the process of developing new funding schemes:

- England is still developing the countryside stewardship higher tier.
- Wales is still developing the sustainable farming scheme, which is due to be complete in 2026.
- The lack of government in Northern Ireland has delayed developments on the environmental farming scheme.
- Scotland is developing a replacement for the common agricultural policy.

For private financing, nature markets were searched from January to August 2024.

For the skills section, we summarised the findings of the following reports:

- CIEEM (2023) Opening Up Vocational Pathways into Nature-based Green Jobs. https://cieem.net/wp-content/uploads/2023/11/Vocational-

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- Forestry Workforce Research August 2021 Forestry Skills Forum: https://www.confor.org.uk/media/2678188/forestry-workforce-research-final-report-august-2021.pdf
- The Green Recovery Challenge Fund: Ancient Woods and Trees- Delivering Landscape Recovery and Ecological Resilience. Final project Evaluation. (2022) Woodland Trust and National Trust.
- ICF and CIEEM position paper – Management of Existing UK Woodlands: An Opportunity for Green Prosperity (Aug 2023)
- Forestry workforce in Scotland prepared on behalf of Lantra and the Scottish Forum. Perth.
- RFS (2017) A forestry skills study for England and Wales. RDI Associates LtD. Ripon.
- Bowditch et al (2022) Finding Forestry:
- Creating a space that enables growth, communication and connection for the forestry sector in the UK. Forestry report prepared for Forestry Commission England.

Results

Available grant schemes for trees and woods

A total of 50 grant schemes were identified through the search (see full findings in tables in Appendix). They do not group neatly into specific categories but grouping them broadly shows:

- Seven UK-wide schemes were identified, the majority of which (six) are Woodland Trust planting schemes and one planting scheme from the Tree Council.
- In England there are currently four grants available for woodland creation, one for woodland management and one for restocking after clearance for tree pests and diseases. In addition, there are the countryside stewardship schemes in which higher tier schemes are not currently open for application and the mid-tier schemes are being merged into other grant programmes and may disappear. These schemes include 27 options around specific elements of management and protection of woodland and trees; and the ELM landscape recovery schemes which are bespoke agreements.
- In Northern Ireland, five grant schemes are available, four of which are for creation, and one for ash dieback.
- In Scotland 11 schemes are available of which four are for creation (two of which are agroforestry), two for management, one for tree health and one for co-operation. The agri-environment climate scheme has five funding streams available for creation and management of hedgerows, scrub, and ancient woods pasture.
- Five are for schemes available in Wales of which three are for creation and two are farm specific.
- 14 private funding options were identified in the search.

Funding awarded

It is possible to find some information on what funding schemes have been utilised and how much has been spent, but it is not always easy to find detailed figures. Here we report information that could be located.

UK

£110.6 million was paid in grants for forestry by the Forestry Commission, Scottish Forestry, Welsh Government and Forest Service Northern Ireland in 2023/24 (Forest Research 2024). Table 1 breaks this down by country.

Table 1 Forest grants paid in each of the four countries in 2023/24 and % increase/decrease compared to the previous year (Forest Research 2024).

England	Scotland	Northern Ireland	Wales
£42.6million (42% increase)	£59.9 million (45% increase)	£5.3 million (10% decrease)	£5.3 million (43% decrease)

According to Forest Research (2024), the total grant money paid in Great Britain has fluctuated over recent years, with levels often dipping around the times that new grant schemes are introduced, followed by a sharp recovery.

England

Forestry Commission England reports in its annual forest statistics, the area of new woodland planted by scheme (Table 2)

Table 2 Area of woodland planted in England 2023-24, excluding trees outside of woods which total 5,529 hectares and 7,091,000 trees.

New planting by type of support	Area of woodland planted 2023-24 (hectares)	Area of woodland planted 2023-24 (number of trees)
Government supported		
Countryside Stewardship woodland	32	55,000
England Woodland Creation Offer	1,648	2,574,000
High Speed 2 Woodland Fund	7	14,500
Forestry England	170	518,000
Countryside Stewardship: other tree planting options	69	111,000
Environment Agency	91	122,000
Northern Forest	238	226,000
National Forest Company	101	83,000
Community Forests	1660	2,225,000
Woodland Creation Partnerships	59	119,000
Green Recovery Challenge Fund	89	80,000
Sub-total Government supported	4,164	6,127,000
Other support		
Woodland Carbon Guarantee	187	154,000
Woodland Trust	197	264,000
Sub-total	383	418,000
Total woodland	4,547	6,545,000

Northern Ireland

In 2023/24, 433ha of woodland was planted under Forest Service grant schemes (Forest Service Annual Report). There are no grants available for ancient woodland restoration currently available in Northern Ireland.

Scotland

The Forestry Grant Scheme updated its figures in May 2024. Since 2015:

- Scottish Forestry has approved **£408 million** on forestry schemes.
- **£352 million** of this was spent on woodland creation (3082 schemes), with the majority spent on commercial conifer plantation (£200 million). Natural

regeneration was also supported (£2.9 million), and £40,000 was spent on agroforestry.

- Around **£9 million** was spent on improving existing woodland habitats for wildlife (including ancient woodland restoration) through 146 schemes. It is difficult to show where this is, and it may have mostly gone on SSSIs where there is legal requirement.

Wales

Table 3 Planting activities in confirmed as planted between 01/04/2023 and 31/03/2024 (Natural Resources Wales, pers comms).

Scheme	Option	Area (ha)	Trees	Conifer/ Broadleaf
Glastir woodland creation				
	803 - enhanced mixed woodland	38.48	96,200	75% / 25%
Small grants - woodland creation				
	SW01 - native biodiversity 1100	5.61	6,171	0% / 100%
	SW02 - native biodiversity 1600	6.89	11,024	0% / 100%
	SW03 - native shelterwood	7.37	18,425	0% / 100%
	SW04 - red squirrel 1600	0.77	1,232	30% / 70%
	SW09 - shelterwood (productive)	2.58	6,450	45% / 55%
	SW10 - wet woodland and streamside 1100	0.12	132	0% / 100%
	SW11 - wet woodland and streamside 1600	0.92	1,472	0% / 100%
	SW12 - wood fuel (productive)	12.98	32,450	45% / 55%
Woodland creation grant				
	P002 - native woodland - biodiversity 1600	41.59	66,544	0% / 100%
	P003 - native woodland - carbon	12.32	30,800	0% / 100%
	P004 - enhanced mixed woodland	190.04	475,100	75% / 25%
	P005 - native woodland - biodiversity 1100	3.21	3,531	0% / 100%
Totals		322.88	749,531	

The Woodland Trust

The Woodland Trust offers several schemes for landowners to help them plant more trees. Data available shows that for these schemes:

MOREwoods has created 1737.3ha of new woodland (2315104 trees) since 2020 and MOREhedges has created 467,244m of hedgerow (89,260 trees outside woods and 2,336,220 shrubs) since 2020. During this period there have been 8,118 applicants for both these schemes.

Table 4: Trees for your Farm: The free trees scheme has had 35,000 applications since its inception and has planted 5829,330 trees.

Year	Number of trees planted	Number of schemes
2021/22	34,802	31
2022/23	60953	26
2023/24	47156	22
2024/25	54,893	28

Sector skills

In addition to the growing need to increase the capacity of the skilled green jobs in the UK to help combat the impact of climate change and nature recovery, there is a circular benefit of this through long-term social and economic benefits of a wider green economy that meets other societal needs such as being a source of secure employment, often in localities where few opportunities exist.

The Department for Environment, Food and Rural Affairs (DEFRA) announced it was collaborating with the Department of Education on a skills gap plan to identify the workforce shortages standing in the way of biodiversity targets. However, there is currently no equivalent plan to quantify this for meeting the much broader aim of net zero emissions by 2050. This skills gap plan report is yet to be published, but some other reports which look at areas of the sector are available which we summarise here.

In 2023, the Chartered Institute of Ecology and Environmental Management (CIEEM) commissioned a report to understand the skills gaps in green jobs (CIEEM, 2023). The report's key findings are:

- There is a capacity crisis and skills gap in the sector.
- There is an overreliance on volunteering.
- Jobs in ecology are unappealing compared to other sectors.
- There is confusion about vocational qualifications and entry.
- The capacity crisis is unquantified.
- The sector is looking for leadership.

Despite a community consensus on there being a capacity crisis, there is currently no solid data to quantify the extent of the problem. There is also no robust quantification of the future demand for skills that will be needed to tackle the climate and biodiversity crises.

Within the forestry and woodland management sector, a number of reports are available that look at the workforce and skills within the sector.

In 2021 The Institute of Chartered Foresters (ICF) sent a position paper to ministers across the UK detailing concerns about the skills shortage in the forestry sector that it believes put climate targets at risk (ICF 2021). The ICF emphasized the need for urgent action to avoid short and long-term consequences of an understaffed and under-skilled workforce. A lack of skills can lead to poorly planted and managed woodlands; for example, 30% of new street trees have been reported as dying within the first few years, often due to unskilled planting. In addition, a survey by the Royal Forestry Society on barriers to woodland creation highlighted the lack of access to professional advice and availability of skilled workers as one of those key barriers, stressing the need for a workforce (RFS, 2019).

The UK has set a number of targets to increase woodland cover: increasing woodland cover to 16.5% by 2050, increasing the area of woodland in management to 75% by 2040 and planting targets of 30,000 hectares a year by 2025 which are currently failing to be met. Recent research has shown there is a significant gap between the number of people needed to meet targets and the number of people joining the sector. The estimated figures needed are between a 32% and 72% increase in Scotland from 2017 to 2027 and 63% to 86% in England and Wales by 2030 (The Forestry Workforce Research (FWR); Lantra 2017). This skills deficit is particularly acute in cities where planting schemes and the services urban trees can provide depend on a small number of local authority tree officers. In addition, the majority of tree nursery staff used to come from Europe, but the EU exit has discouraged workers coming from overseas. There is also concern about the skills and knowledge gaps in incorporating trees on farms. Farms will need to play a big part in tree planting and management as a key part of the transition to net zero. This will require advice from professional foresters to do so. A report from 2021 shows that we could lose 20% of the workforce to retirement by 2030 (Forestry Workforce Research report 2021).

According to the Forestry Workforce Report (2021), the main reason cited by employers for unfilled vacancies is a “lack of skills/experience”. This is leading to inexperienced staff being hired who are dependent on trying to get training on the job. This has highlighted the need for more structured training programmes such as graduate schemes and apprenticeships. The lack of young people coming into the sector is due to preconceptions (low pay, hard working conditions) or lack of knowledge about the sector, as well as challenges around the provision of courses such as them being few in number or the rural location of colleges (FWR 2021). In addition, there are new and emerging skills required within the forestry and woodland management sector in order to meet new challenges faced by compounding threats. The lack of investment in life-long learning to incorporate new skills is problematic. It should also be noted that there is a severe lack of diversity in terms of gender, ethnicity, age etc across the sector that reduces the breadth and diversity of skills, knowledge and experience within the sector.

Research commissioned by the Woodland Trust in 2018 for a Green Recovery Challenge Fund (GRCF) project showed a significant lack of skills and knowledge within the forestry, land management and conservation sectors in the field of ancient woodland restoration and management. These gaps include being able to:

- Identify ancient woodland.

- Understand its qualities and threats and the appropriate actions needed.
- Carry out a site assessment and draw up and enact a restoration management plan.
- Monitor progress.

The report found that where training does exist, it is rarely UK wide, lacks a consistent approach and may not be based on current best practice guidance.

Those contracted to do some of the work for the GRCF project found that they often did not have the skills in-house to complete the work and needed to upskill their current workforce or hire people who already have these skills. However, it was noted that recruiting people with the right skills was very difficult for a number of organisations.

There is a clear need to radically improve the current state of the workforce in forestry and environmental sectors.

Defra's England Trees Action Plan (ETAP) acknowledges the skills shortage but there has been little funding for training in recent years. Scotland's Forestry Strategy 2019–29 makes only a brief reference to supporting education and skills provision and the Welsh Government's forestry strategy recognises the need for increasing the number of people entering the sector. The Green Growth agenda in Northern Ireland has given forestry a high profile but shows little acknowledgement of skills.

The Forestry Skills Forums have long been highlighting these challenges, for example in the Forestry Skills Study of 2017 in England and Wales and the Scottish Skills Action Plan 2020. However, very little has improved in recent years. The Institute for Chartered Foresters is currently working with key forestry organisations in England to produce a 10-year plan for the development of the right skills to support current and anticipated needs.

Discussion

Given the dependence on nature for all, investment in its protection and restoration is vital and urgent. In the UK, the current grant schemes available have a heavy bias towards new woodland creation, and whilst this is an important part of nature recovery, if we only focus on this and not on protecting and improving current woodlands, we risk losing some of the most vital parts of these ecosystems.

Reviewing the current funding available, some key gaps and issues were identified:

- Management. As well as the number of schemes available for creation outweighing those for management, the incentives for creating new woods are stronger than for managing existing woods. For example, in England creation supplements are paid for the ecosystem service delivered (e.g. biodiversity) on top of income foregone and costs. For management and restoration, only income foregone and costs are paid, and this is based on an average so only works on easier-to-restore woods. Private investment is also likely to focus on creation due to the obvious carbon offset benefit. The conservation sector needs to support more extensive natural capital investments and prioritise efforts to better quantify benefits from improved management etc.
- Plantation on ancient woodland sites (PAWS) restoration. There is currently

no support for PAWS restoration in Wales or NI currently and the support in Scotland is very limited.

- Support for collaboration. Collaboration is vital in many areas for effective landscape restoration. This is particularly important for example, in relation to deer/grey squirrel management as this is something better done at landscape scale rather than by individual land managers.
- Trees outside woods. There is currently very little support for managing trees outside woods.
- The Tree Health Grant in England pays for re-stocking ancient woodland sites with non-natives (albeit at a lower payment rate). Given the targets to restore PAWS (see ancient woodland restoration chapter, this report) this seems counterintuitive to meeting policy targets.

In addition to these noticeable gaps, advice also seems to be lacking for some of the schemes. This is problematic if a landowner wishes to undertake creation or restoration work on their land but can't understand how to do so.

It is clear there are not enough people with the right skills to support current and anticipated needs in the forestry sector and meet biodiversity and climate targets. Support is needed to improve training and education from school through to continued professional development to keep ahead of emerging skills needed. There is also the opportunity to provide jobs for communities that lack employment as well as providing opportunities for young entrants and other under-represented groups in the industry.

Conclusion

The benefits woodlands provide in terms of biodiversity, recreation, ecosystem services, health and wellbeing, and increasing resilience to climate change are delivered much more effectively when woodlands are appropriately managed.

Reviewing the current grant schemes and training shows that whilst support for creation of new woodlands is available, management and restoration of existing woodlands does not have appropriate funding associated or skilled workforce required. This leaves landowners with little knowledge, support and incentive to manage their woodlands effectively.

Nature recovery requires meaningful long-term investment and support from governments. For trees and woods, long-term targets have been adopted but only woodland creation appears to be the focus of public investment. With a challenging financial period expected, maintaining existing levels of public investment is essential while extending its objectives to managing and restoring woodland as well as planting. Well-regulated ecosystem service markets have great potential, and the Woodland Trust should engage in their development. There is potentially a need for all parties to work together to support the development of functioning and accessible natural capital markets based on robust evidence and good governance. This might allow public funds to be targeted to areas which are not naturally supported by a natural capital market approach.

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Provenance choice for conservation in a changing world

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Abstract

Trees and shrubs face considerable challenges in adapting to the impacts of climate change as their long lifespans equate to slow rates of population change and consequently of evolutionary processes. Climate projections of generally wetter winters, drier summers and higher frequency of extreme events have raised significant debate about whether UK native species have the ability to adapt naturally to these new conditions, or if they may need intervention to supplement populations with trees from areas that match future climates at creation sites, under the assumption that the latter would be pre-adapted to the forecasted future conditions.

Although individual trees can tolerate a degree of change within their lifetime in their environment, resilience is most likely to come from adaptive change at the population level. This will mean a shift in the genetic composition of a population such that the overall mean fitness of the population tracks the changing environment. Under the right scenarios, it is possible for such adaptation to happen at pace.

When considering the role of trees in a changing world for conservation objectives, management practices that support or enhance natural processes and enable trees to harness their evolutionary potential to adapt to climate change should be prioritised.

Additional adaptive or transformative woodland management practices, such as assisted geneflow (transporting tree seed or material from within the species native range) and assisted migration (importing tree seed or material from outside the species native range, ideally from areas that match climatically either now or in the future), may play a role in mitigating the impacts of climate change, but only in specific situations and when carefully researched and managed to limit potential detrimental ecological consequences.

Considering the uncertainties and challenges, ongoing research and multidisciplinary approaches are essential for developing effective strategies to ensure the survival and health of native woodland ecosystems in a changing climate.

Introduction

Given the magnitude and uncertainty of climate change alongside compounding threats from invasive pests and diseases, clarity on the choice of provenance (geographic location of origin) of seed or saplings for woodland creation with conservation objectives is important. Clarity will ensure that whilst we maintain genetic diversity in our tree populations and support their ability to undergo adaptive change, we don't undermine these processes through introduction of inappropriate material. If we can do this, we will maximise the long-term resilience of our tree and shrub species to environmental changes, and ensure they continue to provide habitat for woodland-associated native species, along with all the other public benefits they provide.

Over the coming century, woods are forecast to face broadly warmer, wetter winters and hotter, drier summers, in combination with an increase in extreme weather events (IPCC, 2023). At the extremes of climate change forecasts, where minimal action is taken by society to mitigate, there is the possibility of novel climates for which there is no current analogue, and these may be associated with no-analogue ecological communities. In worst case scenarios, this could lead to 'discontinuity' where past experience and current knowledge ceases to be a useful guide for future problem solving (Vaillant, 2023; Aitken, 2024). Such conditions would have profound implications for woodland ecosystems and therefore raise important challenges for everyone involved in creating effective adaptation plans (Baumbach et al., 2019).

For the purposes of managing genetic resources, a tree 'population' is viewed as being a group of similarly adapted individuals of the same species living within a particular geographic area and is typically known as a provenance. As tree populations usually evolve slowly, they may face challenges in adapting to sudden changes in climate (Park et al., 2014). Considering this, the choice of provenance when sourcing tree seed is a vital one, as it may influence how successfully trees survive, grow and reproduce over their lifetimes, and there may be few chances for change in the composition of populations once they are established. The overall aim of provenance selection is to source seed that will be well adapted to the prevailing conditions at the site in which it will grow. It is important to keep in mind that, as adaptation is the outcome of exposure to the climate at the source location over time, plus the wider range of biotic and abiotic environmental conditions at that site, predicting how well a provenance will be suited to a given novel planting site can be complex – it is not solely a climate issue.

There is debate over the 'right' solution and whether trees should be sourced locally or from outside the UK (primarily from more southerly latitudes, assuming a broadly northwards warming trend). However, this debate has been often influenced as much by emotions and values as evidence (Whittet et al., 2019). The drive to seek and rely on simple solutions such as importing trees from alternative climates is not without risk and has the potential to have ecosystem-wide consequences (Aitken, 2024; Whittet et al., 2019). Environmental conditions are changing at large scales and are producing surprising outcomes. For example, unexpected range shifts westwards in European forest plants have been found to be due to changes in nitrogen deposition, rather than tracking climate change north as expected (Sanczuk et al., 2024). We still don't fully understand all the factors which are driving change in ecosystems. In addition, depending on the scenario of emissions and associated societal change and global circulation model used to generate projections, there is a huge amount of variability and associated uncertainty in how climate change will play out, both spatially and through time. This uncertainty has been shown to affect predictions of tree height in Scots pine (Hallingbäck et al., 2021). Currently it is rare for this variability to be taken into account in decision support tools aimed at guiding species choice – they are often based on either a selected set of scenarios and models only, or an ensemble average projection based on a number of models.

Importing seed also neglects the considerable natural capability tree species have to disperse by both pollen and seed (called 'gene flow' in this context), and their consequently high inherent adaptive potential. Tree species have varying abilities to adapt to rapidly changing climate conditions, depending on their life history characteristics, genetic diversity, local environment and stress tolerance.

Key practical solutions that can influence and assist tree adaptation include:

- Supporting natural processes of adaptive change. Tree species typically maintain high levels of genetic diversity and, through high levels of interpopulation gene flow, have strong adaptive capacity governed mainly by the rate at which the population turns over (death of individuals vs. recruitment of new offspring). This high diversity equates to a great potential for adaptation to selection pressures such as climate change (Kremer, Potts and Delzon, 2014). Supporting these processes might include monitoring reproduction, ensuring natural regeneration is occurring by restricting overgrazing, or creating disturbance and, potentially, accelerating the rate of change by gap creation to allow new recruits.
- Assisted gene flow is a conservation strategy that involves intentionally supplementing the genetic diversity in a population by moving individuals between populations. Movement is within the species native range and aims to enhance a population's ability to adapt to changing environmental conditions, including climate change. This strategy can play a role in helping tree species, which are often long-lived and slow to adapt, cope with rapid shifts in climate (Gauzere et al., 2020) particularly where populations are isolated from gene flow or are of extremely small size.
- Assisted migration aims to support adaptation and survival by moving individuals to new areas beyond the species' current native range that match future climate projections (Park, Talbot and Smith, 2018).

In the United Kingdom, current policy on tree and woodland provenance choice emphasises the use of locally adapted and genetically diverse tree populations to enhance the resilience of forests in the face of climate change. UK policies encourage the use of native and local seed sources wherever possible, as trees from local provenances are more likely to be adapted to the local soil, climate and ecosystem conditions. This approach aims to promote biodiversity, ecosystem stability and the natural regeneration of woodlands, while safeguarding the long-term health of tree populations. However, with increasing environmental stress from climate change, pests and diseases, the UK's forestry policy also supports the careful introduction of non-local provenances and even non-native species where appropriate, although this is currently under review. The Forestry Commission (2019) advocates for a "climate-adaptive" approach, where trees from slightly warmer regions are introduced to help forests better withstand future climate scenarios. While these policies encourage a balance between preserving local genetic integrity and ensuring adaptability, they also stress the importance of rigorous research and controlled trials to avoid risks such as outbreeding depression and phenological mismatches. This adaptive strategy is aligned with national goals for woodland expansion and climate resilience under frameworks like the

UK Forestry Standard (UKFS).

Scotland's Forestry Strategy places significant emphasis on the careful selection of tree provenance to enhance forest resilience and sustainability, promoting the use of locally adapted species and provenances to ensure that trees can thrive in Scotland's diverse environmental conditions. This adaptive approach supports long-term resilience by preparing woodlands for changing environmental stresses. In tandem with this, Scotland's forestry policy advocates for rigorous seed sourcing practices, blending genetic diversity from both local and non-local populations to maintain ecological balance while enhancing the ability of forests to cope with future challenges. The approach is designed to ensure sustainable woodland expansion and align with Scotland's climate and biodiversity goals.

The Woodlands for Wales strategy takes a nuanced approach to tree provenance, emphasising the importance of using well-adapted, locally sourced tree species to maintain healthy, resilient woodlands in the face of climate change and other environmental pressures. The strategy prioritises the use of native species and local provenance to support biodiversity, ecosystem function and forest sustainability. Trees grown from locally sourced seeds are generally better adapted to the specific climatic and soil conditions of Welsh landscapes, making them more resilient to environmental stressors such as drought, pests and diseases. However, in recognition of the growing uncertainties linked to climate change, the strategy also allows for the introduction of non-local provenances and species where appropriate. This includes the use of seeds from regions with climates that are similar to those projected for Wales in the future, ensuring that Welsh forests can adapt to changing conditions. The goal is to strike a balance between preserving the genetic integrity and ecological value of local woodlands, while also enhancing their capacity to adapt to future climate variability. In line with this, the strategy advocates for the development of seed sourcing and nursery practices that maintain genetic diversity, ensuring robust tree populations for future generations.

A National Forest Tree Gene Conservation Strategy and Action Plan for Ireland has been proposed which incorporates a 'near-nature" approach, focusing on natural processes to create resilient, sustainable woodlands, ideally on an all-Ireland basis. It prioritises native species and diverse, mixed-age forests that mirror natural ecosystems, encouraging minimal intervention and natural regeneration to enhance biodiversity and ecosystem services. Provenance plays a key role, with an emphasis on using local seeds to preserve genetic integrity and adaptability. While it supports introducing non-local provenances to adapt to future climate conditions, this should be done cautiously to maintain ecological balance and prevent genetic disruptions.

The UK Forest Genetic Resources (FGR) strategy is a framework for improved collaboration and research to understand, conserve and use genetic diversity in native tree species, particularly through well-designed sampling protocols that guide the use of assisted gene flow (Trivedi et al., 2019). There are no national or sub-national tree seed programmes – trade in seed takes place at UK scale in both private and public sectors and is controlled by the Forest Reproductive Material (Great Britain) Regulations 2002 and the Forest Reproductive Material Regulations (Northern Ireland) 2002 (Defra, 2024).

The representation of genetic diversity in seed collections is robust, and there is a growing emphasis on adapting seed sourcing strategies to cope with environmental changes. Efforts around FGR collaboration collectively contribute to the sustainable management and conservation of forest genetic resources in the UK. However, we still have a limited understanding of genetic diversity in native tree species, and better information is required to characterise FGR which will inform conservation, use, management and development (Defra, 2024).

Here, we review the evidence to understand the adaptive potential of native tree species in response to climate change to inform provenance decisions for nature conservation. We explore the extent to which; (i) native woods, trees and shrubs have the potential to adapt to climate change, (ii) whether the magnitude and speed of climate change mean native trees are unable to adapt at sufficient speed, and (iii) what the potential ecological implications and outcomes of using non-local provenance trees may be.

Methods

A scoping literature search was conducted using Web of Science and Google Scholar to find both published peer-reviewed literature and grey literature. Search string used was 'forest*' OR 'wood*' OR 'tree*' AND 'provenance*' OR 'genetic' OR 'evolution' OR 'seed' OR 'adapt*' OR 'climate 'AND 'UK' OR "*Britain' OR 'Scotland' OR 'Wales' OR 'England' OR '*Ireland'.

Reference lists were checked to see if they contained any additional relevant studies and studies from outside of the UK were included if they fell into similar climatic and ecological conditions.

Applied Ecology Resources was also checked for relevant literature, as were several organisations' websites including Forest Research, Kew, CEH, Met Office, Natural England, Natural Resources Wales and Nature Scot.

Finally, references suggested by reviewers were also incorporated.

Results

What is the potential for native trees in the UK to adapt to climate change?

Tree species, despite being long-lived and immobile organisms, possess several unique evolutionary characteristics that enable them to adapt to changing environments over time (Cavers and Cottrell, 2015; Geburek and Myking, 2018; Whittet et al., 2019). Characteristics such as large population size, long distance gene flow (via seed and pollen dispersal), phenotypic plasticity and high reproductive output, all contribute to their ability to maintain genetic diversity and resilience in the face of environmental challenges such as climate change, habitat fragmentation and disease. Large tree populations with high genetic diversity enhance resilience, promoting local adaptation and faster evolutionary responses through gene flow (Pluess et al., 2016). Long-distance gene flow (seed dispersal), which is especially effective in wind-dispersed species, spreads genetic diversity, and this raises the probability of species survival in changing climates (Kremer et al., 2012). High fecundity (the ability to produce lots of offspring) in trees also fosters genetic variability, aiding rapid adaptation. This is seen across temperate and boreal species

(Westergren et al., 2023). Overall, the balance between gene flow and natural selection (Cavers and Cottrell, 2015; Whittet et al., 2019) mean that UK tree populations maintain high levels of genetic diversity for natural selection to act upon to initiate an adaptive response. Increased environmental stress can accelerate the process of natural selection, as more individuals are removed from the population and those with traits suited to new conditions are given the opportunity to thrive (Westergren et al., 2023). In addition to this population level adaptation, individual trees' phenotypic plasticity also allows them to adjust growth, physiology and reproduction in response to environmental changes over their lifetimes (Geburek and Myking, 2018).

Maintenance of high levels of genetic diversity provides an essential foundation for adaptation to rapid climate changes (Kijowska-Oberc et al., 2020). Woodlands in northern temperate regions are species poor in a global context, so the genetic variation within species is particularly important (Cavers and Cottrell, 2015). Given that the UK is at the northern limit of the natural range for a number of species, it is thought to contain unique elements of natural variation worthy of conservation (Trivedi and Kallow, 2017). There is substantial within-species genetic diversity in several important British trees (Cavers and Cottrell, 2015), indicating a high capacity for tree populations in the UK to adapt to climate change. Whittet et al. (2019) conducted an extensive review of adaptive genetic variation in English tree species, providing an overview of the current knowledge.

Their findings include:

- Eleven species studied: ash, silver birch, Scots pine, oaks, downy birch, alder, hawthorn, sycamore, beech and rowan.
- Genetic diversity: Tree species hold a larger proportion of their genetic variation within populations than among them. This means that most populations contain broadly the same amount of genetic diversity within them, with only a few small differences (Figure1). This suggests a strong capacity for adaptation through natural selection, as there are always individuals within a population capable of evolving to cope with environmental changes. It also indicates a high level of gene movement (via seed & pollen dispersal) between populations. If inter-population variation were higher, it would imply that some populations could thrive while others might completely fail (Whittet et al., 2019), and that there were low levels of gene movement to provide new diversity. For example, in Northern Ireland, black alder populations display greater genetic diversity within populations than between them (Beatty et al., 2015).
- Species-specific variation: The spatial scale of genetic diversity varies across species, determined by characteristics of their life history such as lifespan, seed and pollen dispersal mechanism, levels of reproductive output, and sexual and mating systems.
- Site vs. provenance: The characteristics of the growing site (such as altitude, topography and soil type) often have a greater impact on tree performance than provenance, making careful site selection equally, if not more, important.
- Unexpected results: Stressful growing environments, like high altitudes, can sometimes produce outcomes that defy expectations.

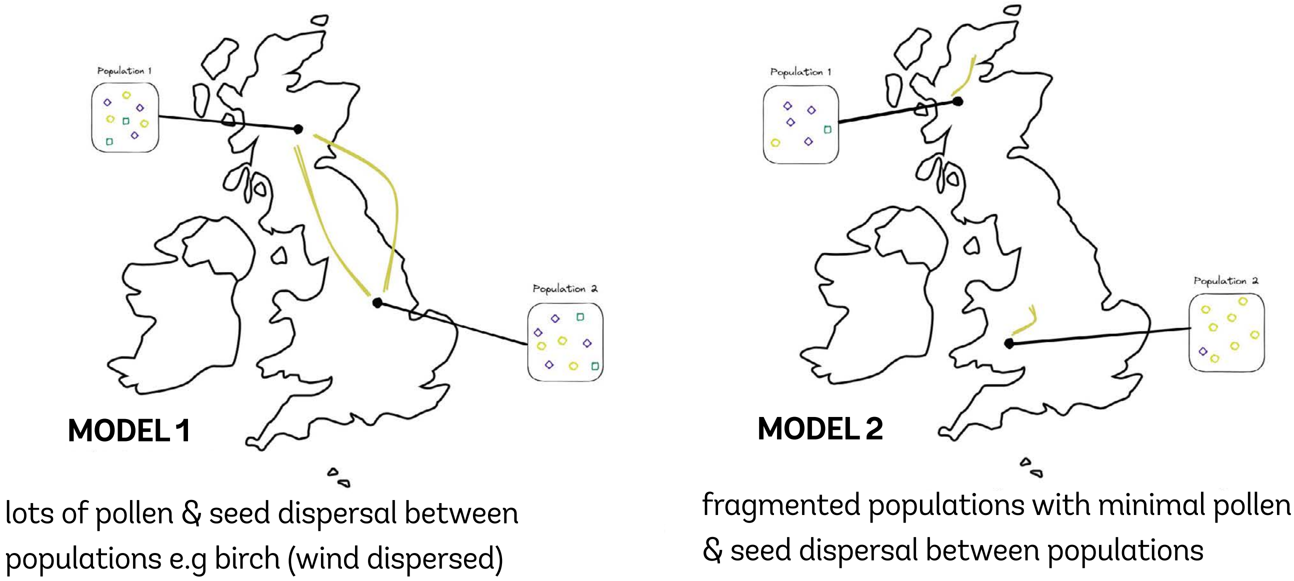

Figure 1: Generally, for our UK native species, a greater amount of genetic diversity is partitioned within populations (model 1), as opposed to between them (model 2). This means that where model 1 applies, there will always be some individuals within populations which can adapt. If model 2 were the case, populations would be very different from each other and more similar within themselves – so would be more at risk from new threats wiping them out.

Will the magnitude and speed of change mean this ability to adapt is pushed past its limit?

The rapid pace of climate change poses significant challenges for native tree species in the UK. With predictions of increased temperatures, altered precipitation patterns and more frequent extreme weather events, there is growing concern around whether native trees can adapt quickly enough to these changes.

If the long-lived nature of trees means populations are slow to change in genetic composition, this may prevent species from harnessing their adaptive potential fast enough in response to climate change (Cavers and Cottrell, 2015). There is limited research to demonstrate whether or not tree species will be pushed beyond their adaptive limit. Whilst studies have begun to apply species distribution modelling to explore range shifts and the ability of tree species to migrate quickly enough to track the changing climate, these models have generally not accounted for genetic variation and the possibility of adaptive responses in combination with climatic data (Wessely et al., 2024, Whittet et al., 2019, Yu et al., 2021). The potential for rapid genetic response is seen in cases where regeneration does occur, such as in oaks, where research has demonstrated rapid evolutionary responses to selection pressures generated by sharp climatic changes and that rapid evolution is probably already underway (Caignard et al., 2024).

While local adaptation, genetic differentiation and phenotypic plasticity are crucial for survival, and evolutionary mechanisms may allow trees to adapt relatively quickly, the pace of adaptation may still lag behind environmental shifts, necessitating strategies such as assisted gene flow (Cortés, Restrepo-Montoya and Bedoya-Canas, 2020). This is gaining attention as a strategy to help tree species adapt by relocating trees (via seed or genetic material) from locations where current climates match those emerging elsewhere in their native range. This approach requires careful consideration of the potential ecological and biosecurity risks and consequences (Park, Talbot and Smith, 2018), and there is currently limited research on this in a UK context.

Current tools to support decision making on non-local provenance and assisted gene flow are useful but have limitations. They are often built using long-term average data for a few climate variables, whereas the key selective pressures may be infrequent events e.g. a one in 20-year spring cold event. In the north especially, this adaptation to site may manifest as conservative behaviour (e.g. late bud burst /early dormancy, small stature, early reproductive maturity) (Vander Mijnsbrugge et al., 2016). Models also often neglect biotic interactions and soil characteristics. There is overwhelming evidence that local adaptation in trees occurs not only to climate but to many other aspects of the environment, particularly soils and soil biota, herbivores and pathogens (Ennos et al., 2019). Decision support tools that include detailed site matching, including a range of other variables along with climate e.g. ESC (Pyatt, Ray & Fletcher, 2001) can be refined further (Whittet et al., 2019).

What are the potential outcomes when using genetic material from non-local provenances?

A growing number of studies recommend the use of seed from mixed sources to anticipate the potential impacts of climate change (Thomas et al., 2014), however, the introduction of seeds or trees from wider provenances, either via assisted gene flow (across the native range), or via assisted migration (beyond the native range), raises concerns about local adaptation and the impact on dependent biodiversity communities due to the unpredictable risk of disrupting natural patterns of within species biodiversity and their ecological communities (Bucharova et al., 2019). More research is needed to understand these potential consequences to consider assisted migration as an appropriate strategy (Twardek et al., 2023).

Sourcing seed from currently warmer climates can increase growth trait values (Whittet et al., 2019), but conversely there may be detrimental impacts and risks associated with assisted migration. Maladaptation (reduced performance in traits including survival, growth and increased susceptibility to pests and disease) is a potential risk when using non-local provenances, especially under changing climatic conditions. A cautious approach that incorporates climate predictions and a wide range of genetic diversity can help mitigate these risks (Gellie et al., 2016). Research indicates that using a more diverse mix of seed sources from varying climatic conditions may mitigate some risks of maladaptation. However, this requires careful selection to avoid outbreeding depression and other complications (Frankham et al., 2017).

Translocated seed has also sometimes been shown to be initially ill adapted to the current climate, but this is species dependent and there are no one-size-fits-all patterns. For example, with northwards movement of alder buckthorn and blackthorn there is an increased risk of exposure of buds and foliage to late spring and early autumn frosts compared with locally adapted stock, as the timings of leaf and bud emergence and senescence may not be phased to avoid frost, due to past adaptation to a longer growing season (Vander

Mijnsbrugge et al., 2016; Vander Mijnsbrugge, Turcsán and Michiels, 2016). Frost damage may reduce the chances of establishment of translocated seedlings or adversely affect the form of trees if leading shoots are killed by frost, causing forking and resulting in lower fitness and competitiveness of trees later in life, as shown for temperate deciduous trees (Vitasse, Lenz and Körner, 2014) and specifically ash in the UK (Rosique-Esplugas et al., 2022). Thus, maladaptation both to initial climate and to other important aspects of the environment is likely to result in lower initial survival of provenances translocated in anticipation of climate change.

Maladaptation may not become apparent for several years. Exposure to relatively infrequent events such as harsh spring frosts after mild winters, rare or seasonal flooding events, high winds, and droughts (Benomar et al., 2022; Casmey, 2022) may cause problems for established trees which are not adapted to such conditions (Whittet et al., 2019).

Discussion

Provenance choice is an important consideration for climate change adaptation because it can have an impact on how well tree populations will survive and thrive under changing environmental conditions. Given the complexity of the combined climate and nature crises, we need to be aware of the danger of overly simple solutions, e.g. "if only we plant more trees" (Aitken, 2024) or "if only we bring in trees from x location". Broadly, the aim of provenance choice is to provide well adapted trees, to make woodland ecosystems resilient to change, and this can be interpreted in different ways. As highlighted by Cavers & Cottrell (2015) distinguishing between whether the priority is 'tree species resilience' or 'whole ecosystem resilience' is also vitally important when framing choices. In some areas of the UK, woodland communities may be less species-diverse, but their unique composition is inherently important, and therefore their integrity is driven by the ability of those particular tree species to adapt to climate change (Cavers & Cottrell, 2015). To achieve conservation objectives in such species-poor woodland, creating the conditions for evolutionary mechanisms to act via natural processes is essential – this would be a focus on 'species resilience'. In contrast, choosing ecosystem resilience may involve prioritising an overall woodland metric (e.g. woodland cover) or a public benefit (e.g. carbon sequestration) over preserving species composition, risking the loss of species-specific associated biodiversity and ecosystem processes (Cavers and Cottrell, 2015).

If we think of resilience as the ability of a system to resist, adapt or transform in response to change (Millar & Stephenson, 2015; Aitken, 2024) then we can frame the findings within each of these options.

Resist

Protect our ancient and semi-natural native woodlands

Globally, resisting change and choosing species resilience is most relevant for old growth forests (Aitken, 2024). In a UK context this applies to in and around our remnant ancient semi-natural woodlands and ancient and veteran trees. As highlighted in other areas of the report, values play a central role here too, with considerable value placed on the social and cultural aspects of native tree species for their association with natural history, folklore, and the provision of habitats for biodiversity (Defra, 2024). These values, and their role in arguments for the conservation and prioritisation of native tree species, should not be overlooked as they play a powerful role in decision making and vary between stakeholders involved in woodland ecosystems. It's clear that although species poor in a global context, our native woodlands represent irreplicable genetic resources and house a significant amount of genetic diversity from which to work from. However, for managers of ecosystems in our changing world, resisting ecological change even where feasible may require sustained and intensifying efforts (Schuurman et al., 2022) and active measures are likely required to make the most of this capacity. This will require strong protection and good management of our existing woodland resource to improve its condition (see condition and management sections this report).

Collect and preserve genetic material

It will also require actions to collect and preserve genetic material from ancient semi-natural woodlands. The recent State of Forest Genetic Resources (FGR) report has summarised the current state of FGR in the UK (Defra, 2024). In-situ conservation of some tree species currently occurs via a network of Genetic Conservation Units (GCUs), where woodland sites are designated for their naturalness, sufficient population size and assumed genetic diversity. However, there are currently only 15 of these designated for six species, with all of these being relatively new and there currently being no centralised effort to maintain and monitor them (Defra, 2024).

Approaches to ex-situ conservation include seed banking or clonal banking and require quality seed collection practices to sample the current genetic resource in native woodlands. In creation and restoration efforts, sapling cultivation in nurseries is a common practice, though it may hinder natural selection and affect genetic diversity. To mitigate this, careful seed collection practices are essential to ensure genetic diversity in seed lots and tracking of collection sources (Whittet et al., 2016). Combining seeds from both managed and natural populations helps nurseries preserve genetic diversity, while nursery conditions can foster physiological adaptations to improve seedling survival (Turchetto et al., 2016). Seed collection also occurs purely by or for sale to nurseries, or for the creation of seed orchards. There is no national programme for this, with trade occurring publicly and privately and being regulated by Forest Reproductive Material (GB) Regulations 2002 and the Forest Reproductive Material Regulations (Northern Ireland) 2002 (Defra, 2024).

There are challenges to make seed collection and growing as local and bespoke as would be ideal from a genetic perspective. Despite recent targeted efforts, the resource for native trees is limited in practice. The UK National Tree Seed Project aimed to establish multi-provenance seed collections representing the majority of adaptive genetic diversity present in the UK and this database will be made publicly available soon. In England, the Seed Sourcing Grant, and Tree Production Innovation Fund have both financed a number of projects in recent years aimed at increasing the quality and supply of forest genetic resources. The Woodland Trust has been the recipient of two rounds of SSG funding, with this driving activity to assess sites for suitability as seed stands for eight priority species across the estate in England. Although providing a valuable mechanism to do this, it has highlighted the current gap between the aspirations to register more stands, and the high resource required to implement the management needed to bring seed stands into a usable condition. The management need is often associated with providing access, but also with wider poor woodland condition and declining levels of management which threaten to create conditions (e.g. poor light levels) which may be limiting fruiting (see condition and management sections of this report).

Adapt

Adaptation can happen at speed if given the chance

UK native tree populations exhibit a range of evolutionary mechanisms to cope with environmental change, including high levels of genetic diversity and genetic flow, which are essential to enable a rapid adaptive response to selection pressures from dramatic climate shifts. Given the pace of climate change, ideally adaptation needs to occur rapidly within populations across the species' range. Although research in this area is currently limited, under the right conditions, populations can begin to adapt within a single generation. For example, Metheringham et al., (2022) show an adaptive shift in allele frequencies related to resistance to ash dieback in seedlings. This study demonstrates that significant change can be detectable after only a single generation following a major threat. Findings such as this suggest that there is the possibility to achieve species resilience by seeking genetically tolerant varieties of threatened native species.

Furthermore, a study investigating the response of Scots pine to the disease Dothistroma needle blight demonstrates that adaptation is possible across the range of a species (Perry et al., 2016): individual trees with high levels of resistance to the pathogen *Dothistroma septosporum* were found even within otherwise 'susceptible' populations, highlighting their high genetic and phenotypic diversity and suggesting that populations have a high internal capacity to respond and survive. This suggests that there is plenty of gene movement (or migration, via pollen and seed dispersal) occurring naturally between populations, so assistance is likely not required, beyond ensuring that these advantageous genes can enter the population through natural regeneration. These findings demonstrate that tree populations can cope with threats in situ and that it is not necessary to select a single 'well adapted' provenance.

With comprehensive genetic data for just 11 native tree and shrub species (Whittet et al., 2019), it is possible that there is under or un-utilised/latent diversity in populations which won't be realised until new selection pressures arise. It is possible that this means that there may be many genes which play little or no role in fitness in current environments, but that could become important as conditions change. More research is needed to address the scarcity of genomic data for native trees to aid our understanding of their adaptive potential and responses to climate change, but it's important to note that more genomic data won't deliver any information fast enough to change the broad principles of what we currently know.

Tree nursery conditions also have a vital but under-researched role in tree development and adaptation later in life. Not enough is currently known about the impact of their associated climate on the growth and development of seedlings, yet recent studies of Scots pine (Pinus sylvestris) have shown that conditions can have significant effects on measured traits (growth and phenology) later in life (Perry et al., 2024b, 2024a). This has significant implications for consideration of the environments in which we germinate and grow tree and shrub stock and is an important area for further research.

Management actions are required to promote rapid adaption

Trees have been acting on selection pressures and adapting to changing environments for millions of years. However, if we want to achieve particular outcomes in the face of the twin biodiversity and climate crises and mitigate risks of unintended consequences, aiming to work with natural processes may lead to better long-term resilience and conservation outcomes in our woodlands. Doing this will rely, to a large extent, on the inherent variability of tree and shrub species, their dispersal mechanisms, and the recruitment of new genotypes into their populations. Relying on this natural approach may result in a short-term temporary decline of existing generations whilst reproduction and recruitment establishes new, better adapted generations (Cavers and Cottrell, 2015). This presents a challenge given the many social and cultural values held for trees, as evidenced by societal reactions to tree loss from Dutch elm disease and ash dieback. Where enabling natural processes, and potentially allowing more significant short-term losses of trees of more threatened species, there needs to be careful communication with key stakeholders and the public.

However, enhanced management can help to both mitigate the impacts of climate change and aid evolutionary mechanisms. Woods across Great Britain are in poor condition, suffering from neglect and a lack of long-term management (Smart et al., 2024) (see management section of this report). This lack of management has led to reduced levels of natural regeneration, and therefore reduced opportunity for selection of individuals adapted to a changing climate. Barriers to adaptive change in tree populations are predominantly management related – e.g. herbivore pressure, habitat loss etc. rather than any inherent lack of evolutionary capacity (Cavers and Cottrell, 2015). Management strategies should prioritise leveraging existing genetic diversity to enable evolutionary adaptation. New approaches could include increasing the frequency of disturbance through gap creation and conservation grazing (Cavers and Cottrell, 2015) and adopting approaches such as 'disturbance-based management', 'close-to-nature silviculture' (Brang et al. 2014 in Whittet & Quine, 2024) and 'evolution-oriented management' (Lefevre et al. 2014 in Defra, 2024).

Using natural processes for woodland creation

At just 2.5% of land cover, our ancient semi-natural woodland is a depleted and threatened genetic resource (Woodland Trust, 2011). Although sometimes difficult to implement in practice, it is an accepted strategy in landscape ecology to protect, restore and expand areas of core woodland habitat, by encouraging creation directly adjacent to them (Synes et al., 2020; Mancini, Hodgson and Isaac, 2022). Ideally, this expansion should occur via natural processes of seed dispersal and suckering, to enable the natural processes of natural selection and adaptation to operate. There are grants to support this, but they are currently limited to only some UK countries (e.g. England, Scotland) and regions (e.g. Northern Forest).

There may be limits to adaptation, but decision support models need developing further

Whilst adaptation is possible, there may be limits under extreme climate change where the long generation times and low mortality rates of trees may slow adaption at the same time as environmental changes happen faster. There is minimal research on this in a UK context, and there are limitations associated with the methods used to be aware of. The extent and variability of extreme climate change is extremely difficult to predict and model. Many decision-making tools have been developed to inform provenance choice, typically using macroclimate (large-scale) data (Maclean and Early, 2023; Souza Lima, Lenoir and Hylander, 2024). For instance, (Yu et al., 2021) used a dynamic ecosystem model to study 12 tree species in Great Britain and found southern woodlands are increasingly vulnerable to climate change and drought, affecting species like beech and lime. In contrast, northern species such as ash and oak may be less affected. However, the model's accuracy is constrained by limited ecological data and species parameterisation for Europe rather than Britain. It also deals with standing trees only, taking no account of the dynamics of a changing population. Models such as this may overestimate range shifts, as the distribution of many species' changes in response to climate change are likely to be localised and difficult to infer when only coarse-scale microclimate data is available. When fine-scale topography, soil and vegetation data are used, much smaller and more localised shifts are predicted (Maclean and Early, 2023).

Generally, the studies which suggest more catastrophic declines in tree species fail to make use of the genetic data available to deal with the dynamics within species, and this hinders accurate predictions of climate impacts on native trees (Wessely et al., 2024). When these dynamics are considered, for example in field studies of sessile oak (*Quercus petraea*), findings indicate that although climate change is predicted to have some negative impacts on height and survival, populations are less vulnerable than thought as long as the balance between gene flow and adaptation via regeneration is maintained (Sáenz-Romero et al., 2017). New emerging trait-based species distribution models incorporating phenotypic plasticity and local adaptive potential in trees deliver a less alarming message than previous models of distributions based purely on future climates (Benito Garzón, Robson and Hampe, 2019). Another complication that isn't yet captured in models is that climatic tolerance in trees also varies ontogenetically (i.e. throughout their lifespan from seedling to maturity). Long-term provenance trials provide opportunities to measure ontogenetic trait variation linked to climate, providing a vital resource with which to inform interventions (Erlichman et al., 2024) and minimise the risks of perverse outcomes. Overall, we've learned a lot from genomic approaches, but we have to be cautious in our interpretation of them (Aitken, 2024). These approaches rely on models built from the best available big data, but we don't fully understand them, and they aren't ready for application until they have been validated via long-term field trials (Aitken, 2024).

Benefits and risks of assisted gene flow (same species but different genetics)

A step further than enabling natural adaptation would be assisted gene flow (AGF), which is human-mediated action to introduce genetic variation into populations which may be facing fitness declines (Grummer et al., 2022). In a UK context, this could involve using species which are already found in the UK, but that are sourced from non-local provenances from regions with climates that resemble predicted future UK conditions. This is expected by some to help future-proof UK woodlands against increasing temperatures and droughts (Yu et al., 2021), but while non-local provenance trees can have both positive and negative effects on woodland-associated species, the impacts are highly context dependent. The introduction of non-local provenance alleles 'pre-adapted' to emerging climatic conditions could enhance adaptive capacity, genetic diversity, and resilience to climate change in woodland conservation strategies (Boshier et al., 2015). However, for many UK native species, it seems that there is enough existing genetic variation within populations, and that AGF may not be required. Genetic modelling has indicated that while AGF can aid adaptation if enough material is moved, its impact is often small or delayed, especially in large populations (Grummer et al., 2022), so resource may be better focused on enabling natural adaptation.

There has also been little published research on the wider ecological consequences of AGF on woodland ecosystems and biodiversity. Associated risks, such as adaptive or phenological mismatches, hybridisation, or outbreeding depression, must be carefully balanced against any potential benefits and should be tested through controlled provenance trials (Grummer et al., 2022). For example, sourcing oak trees from non-local populations using a climate matching approach demonstrated declining growth performance with increased distance from between provenance and the new site and also showed an increasing phenological mismatch resulting in a decline in gall wasp abundance (Sinclair et al., 2015). Effective management and rigorous risk assessments are essential to mitigate potential negative consequences like this, and to support the conservation of native species.

Transform

More extreme predictions from European scale research predicts decline in many tree species due to the speed and scale of anticipated climate change (Wessely et al., 2024). This increasingly leads to suggestions of assisted migration (moving material from outside species native range), or even the use of new non-native 'novel' species. However, research from Canada has shown that assisted migration has less support from practitioners and publics than assisted gene flow (Hagerman and Kozak, 2021), and novel species are subject to intense debate (we don't consider them in this report).

Limited evidence and higher risks

The evidence for assisted migration as a tool for sustainable forestry is more extensive in comparison to the requirements of native trees and woods for conservation purposes. Although it may have merit in commercial sectors if done within a comprehensive risk assessment framework, the approach has been dismissed for situations where biodiversity conservation is the objective, with there being no evidence for introduction of any non-native species (Ennos et al., 2019). In the context of productive forestry, it may also be the case that gains would be better achieved from selecting improved varieties (i.e. from breeding trials) of native or long naturalised species, without needing to make use of assisted migration.

As well as the aforementioned limitations of the models currently used to explore assisted gene flow, and the uncertainties associated with forecasting future climate change and site-matching future conditions, there are increased ecological risks associated with assisted migration. These are dominated by the risk of introducing invasive pests and diseases, and displacement of native species (Argüelles-Moyao and Galicia, 2024; Michalet et al., 2024) (see tree health, extent and ancient woodland restoration sections of this report). Globalisation over the 21*st* century has been associated with increasing trends in pests and diseases across the globe, and this has had a combined negative effect on forests in the US (causing tree mortality on large scales) in the same region as losses from wildfire (Fei et al., 2019). This is due to the lack of co-evolved defences which develop through thousands of years of co-evolution, as well as 'enemy release' (Leibhold, 2024). This is the phenomenon where species usually have communities, including predators, around them, but they are typically moved without these, so populations can explode in new environments. This may also explain why non-native trees often grow better than native ones, as they are not limited by any relationships with native insects or pathogens (Leibhold, 2024). This effect can be seen particularly in Scotland, where non-native trees are escaping from plantations and into areas prioritised for nature. The effect of this enemy release does erode over time, with non-native trees becoming more vulnerable. Biosecurity practices and regulations are therefore imperative in the woodland and forestry sector regardless of objective.

Choosing different provenances or even novel species would imply choosing 'ecosystem resilience'. An example of this framing of resilience would be allowing or encouraging sycamore (although a naturalised species) as an alternative to ash in response to ash dieback. The risk in this case is particularly high in low nutrient systems in the British uplands, where a replacement of ash with sycamore, although maintaining overall woodland cover, would have significant impacts on light and nutrient levels, with risk of knock-on effects in ecosystem processes. It is increasingly argued that nativeness should not be such a binary concept, and that wider definitions could be developed where species are characterised based on their potential level of risk or possible beneficial functions (Warren, 2007; Lemoine and Svenning, 2022). For the above ash example, nuanced advice has suggested an approach which uses multiple species to provide habitat for ash-associated species, as well as the ecological functions provided by ash (Broome and Mitchell, 2017). This provides a framework for a potential alternative or complementary approach to assessing species choice, with choices being ecologically guided, based on ecological function, as opposed to only considering climate and site conditions.

Conclusions

Provenance trials and genetic studies provide insights into the adaptability and resilience of different tree populations, which is essential for conservation efforts. Whilst many in the forestry sector are now advocating for assisted migration, the evidence to support this as a priority intervention suggests that it is highly context-dependent. The adaptation of tree populations to climate change involves a complex interplay of physiological, genetic, ecological and evolutionary processes. Management practices that incorporate evolutionary understanding and sustain genetic diversity and gene flow are crucial for maintaining the adaptive capacity of native woodland. In some cases, and for some applications, e.g. productive forestry, assisted migration may be a valuable tool.

There is currently limited research on assisted gene flow and assisted migration in a UK context. Any decisions on these strategies carry risks of perverse outcomes if done poorly and need to be informed by extensive research and monitoring (field trials). There is therefore currently insufficient evidence to drive a targeted assisted strategies.

When considering provenance choice for native woodland creation to achieve conservation rather than production outcomes, unless a population is in critical decline, it should be a priority to focus on ensuring the integrity of ecosystem processes and supporting the natural adaptive capacity of tree species, rather than on potentially oversimplified approaches based on uncertain projections. For woodland creation and management for conservation objectives, priority should be given to working with evolutionary mechanisms, primarily through ensuring natural regeneration, and allowing natural selection to create future woodlands with trees comprising individuals selected to thrive in a changed climate. In particular it is essential to ensure that natural processes are given time and space, as it is only by allowing new genetic diversity in that populations can adapt.

Native tree and shrub species have the evolutionary characteristics and population structures to be able to adapt to change, but appropriate management to support natural processes is essential to enable this. Continued research into the genetic and phenotypic mechanisms of adaptation will help develop strategies to mitigate the impacts of climate change on woodland ecosystems, and to optimise strategies for particular species.

Key knowledge gaps

- How do the populations of native UK trees (beyond those important for forestry) vary at a finer scale (i.e. not range wide) (Cavers & Cottrell, 2015)?
- What are the local/fine-scale factors which govern adaptation including climate, soils and pathogens across the UK, and how these factors interact to produce genetic variation within species?
- Detailed genetic characterisation of our existing and future gene conservation units (GCUs) (Defra, 2024).
- Which methods can best support or accelerate natural regeneration: what site management methods will bring new genotypes into the population faster without compromising the existing woodland?
- Do provenances of UK native species differ in terms of their susceptibility to existing or novel pests and diseases (Cavers & Cottrell 2015)? This lack

of evidence contributes to debate on whether the current seed zone system reflects the broad distribution of adaptive variation in all tree species in the UK well enough (Cottrell & O'Hara, 2017).

- Which nursery environments give young trees from different genetic backgrounds the best chance of thriving in future climates?
- What are the ecological outcomes on woodland-associated species of introducing non-local provenances in a UK context?
- In which situations can non-native trees provide beneficial ecological functions? When and where do non-native species become invasive, and what are the effects?

Definitions

Genetic diversity – the variety of differences in the genes within a species.
Individual – a single tree.

Genotype – the genetic makeup of an individual (genes, made up of alleles Aa AA aa etc).

Trait – characteristics that result from genes.

Phenotype – traits or characteristics of an individual, which result from the interaction of its genes (genotype) with the environment.

Population – a group of trees of the same species living together in a certain area. These trees interbreed/reproduce, passing their genetic traits to the next generation.

Provenance – the geographic location of a population of trees.

Natural selection – the process through which the individuals that are best adapted to current local conditions survive, reach maturity and produce offspring. Generally, most intense when trees are at the seedling/sapling stage, when mortality is highest. It causes traits that contribute to adaptedness to increase in frequency in a population and, over time, may cause populations to differentiate from each other (Cavers and Cottrell, 2015).

Gene flow – dispersal of genes via seed, pollen, or vegetative propagules. Tends to counteract the effect of natural selection (Cavers and Cottrell, 2015).

Inbreeding depression – when closely related individuals breed, greater risk of inheriting harmful genes, reducing the populations overall diversity and resilience.

Outbreeding depression – when individuals from different populations breed, and offspring end up with traits that aren't well suited to the environment – again leads to weaker offspring that may struggle to survive or reproduce.

Phenotypic plasticity – the ability of a single genotype to produce. different phenotypes (observable characteristics) when exposed to different environmental conditions. This may provide temporary relief (Grummer et al., 2021) e.g. reducing leaf area and growth rate to tolerate increasing temp and decreasing precipitation.

Adaptation – the result of natural selection acting on within species genetic diversity. It is influenced mainly by the balance between natural selection and gene flow. Under changing conditions, the composition of genotypes in a population may gradually alter, e.g. those that are more tolerant of drought may be favoured as conditions become drier. Local is not fixed but dynamic, resulting from a combination of chance events (mutation, genetic drift and gene flow) and natural selection, acting under a changing biotic and abiotic environment (Cavers and Cottrell, 2015). A population's rate of adaptation is proportional to the extent of genetic variation it harbours" (Grummer et al., 2021).

Assisted gene flow – human mediated, intentional translocation of individuals within a species range to facilitate adaptation. A way of introducing more genetic variation.

Assisted migration – human mediated, intentional translocation of individuals from outside native range.

Provenance trial – trees from different provenances are grown in a common environment. Therefore differences among provenances can be inferred to be due to their genotype. Results of provenance tests are used to help guide decisions about deployment of seed sources to planting sites (Whittet et al., 2019). A provenance trial tests for genetic variation in phenotype among provenances (how much traits vary between genotypes if the environment is the same for all). Multisite provenance trials (a provenance trial replicated across sites) will test both genetic variation in phenotype and phenotypic plasticity (how much the traits of a given genotype vary across different environments). The latter could also look for genetic variation in plasticity (how much phenotypic plasticity varies among genotypes).

In-situ conservation – protecting species in their natural habitats e.g. GCUs.

Ex-situ conservation – moving species (or seed in this case) to protected places e.g. seed banks or places e.g. seed orchards).

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Creation and expansion

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Introduction

Across the UK, there are ambitious and challenging targets intended to increase the extent of native woods and trees to tackle the climate and nature crises and create landscapes rich in native woods and trees, for people and wildlife.

The expansion and intensification of agriculture and urban infrastructure and development over the last century has reduced and fragmented our wooded habitats. Woodland cover in the UK is currently around 3.2 million hectares, roughly 13.2% of the total land area. While this is almost triple the woodland extent 100 years ago, priorities for expansion and resulting woodland composition have changed over time, initially being focused on increasing the UK's timber reserve, but becoming more multi-functional through time, with increased support for native woodland for biodiversity and other public benefits. However, recent resurgence of concerns around timber security is possibly driving the balance back the other way. The rate of expansion has also slowed in recent years, with a 1% increase over the last two decades. In order to meet net zero and nature recovery targets, it is vital that we find ways to address the underlying reasons for this lack of progress.

Although ambitious and urgent, targets are also non-specific and open to varied interpretation and have rarely been met on an annual basis. A proliferation of opportunity mapping work has been carried out at various scales, demonstrating that there is technically enough suitable land for woodland creation in the UK. Although priorities and methods vary, a common feature of this type of work is to ensure that valued existing land uses, such as high-quality agricultural land and other nature designations, are protected and not compromised by creation. Alongside this, incentives (mostly in the form of various government grants, but also via private or charitable organisations) have been available for decades, making creation an attractive option financially for some land managers. Despite this, it's clear that knowledge of theoretically available land and grant-driven creation is not achieving the required change. Attempts to drive change at the landscape scale are challenging, especially as developments in ecological restoration thinking and policy over recent decades increasingly see people as central to ecosystems and aim to drive true participation in environmental decision making. It's increasingly argued that approaches to do this need to be built around non-economic values which focus on people's preferences, principles, and responsibilities towards the environment. This is easier said than done, with attempts to integrate the environment into decision making to date primarily being driven by quantitative mapping and economic valuation of the benefits expected to be delivered by creation. It's been shown that institutional and governance concerns, along with set social norms, not economics, are the most severe obstacles to implementation of ecological restoration (Sayer et al., 2013). In other words, land use change (woodland creation in this case) has been treated as either a biophysical (land suitability) or economic issue, being seen as a predictable and rational process, rather than the social (or negotiation) process between groups of people with different values, that it often is.

Whilst other barriers such as access to advice, lack of knowledge, time and cost are important to consider (Staddon et al., 2021, funding and skills gaps section this report) and there is a useful body of knowledge understanding woodland creation and management through other social and psychological frameworks (Ambrose-Oji et al., 2019) we focus here on values to highlight how these can aid in our understanding and provide a lens through which to understand these other factors.

Research into woodland ownership and management has often been explored through segmentation and typologies (Urquhart and Courtney, 2011; Eves et al., 2013; Ambrose-Oji, 2019). These typologies are intended to reflect the range of woodland owners and agree on a set of landowner types, with groups typically identified as having either economic, production or amenity and/or conservation concerns, although some are referred to as either multifunctional or multi-objective owners (Ambrose-Oji, 2019; Ficko et al., 2019). The intention is that by better understanding and characterising different kinds of land managers, interventions can then be tweaked to target different groups. These typologies have given us enough of an understanding of the kinds of land managers who are already likely to engage with woodland creation (e.g. younger farmers, new entrants, those with higher levels of education, those wanting to leave a legacy, and those with previous experience of creation or other forms of diversification), and we can target these via policy or advice (Staddon et al., 2021). However, these typologies have typically been quantitative and have focused on objectives and socio-demographic characteristics, masking variation within each landowner or manager type (Staddon et al., 2021). Further development of typologies is unlikely to drive change. What's needed then, is a deeper exploration of the factors driving behaviour. Attitudes, beliefs, world views have all been noted for their importance to decision-making (Staddon et al., 2021) and are underpinned by values (Dietz et al., 2005). Values are a fundamental driver of behaviour and play an influential role in shaping policy, schemes and mechanisms (Chapman et al., 2019). As such they are particularly important to understand in relation to land -managers' behaviour and the uptake of schemes and mechanisms for woodland creation. This growing recognition is reflected in the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) values framework, which emphasises the plurality of values that people hold relating to the environment and how these influence their behaviour (Diaz et al., 2015).

In this chapter we set out the current progress towards reaching creation targets in the UK and explore the role that people's values could play in land use change decision-making and achieving these targets.

Methods

Rate of creation against targets

Analysis was carried out in R version 4.2.3 "Shortstop Beagle", using packages tidyverse and ggplot2. The latest Forestry Statistics data for creation was downloaded from https://www.forestresearch.gov.uk/tools-and-resources/

statistics/data-downloads/ and country level data was collated into a single dataset. The data was filtered to years 2020 – 2024 and summarised to show the mean total annual area of woodland created over that period, along with the standard deviation, per country. This data was then plotted against the Committee on Climate recommended annual targets required to meet net zero (Committee on Climate Change 2020).

Evidence review

Literature searches were carried out on Web of Science, Google Scholar using the following search terms. Country specific governmental websites were also searched for relevant grey literature.

(Wood* OR forest* OR tree*) AND (Value* OR "Ecosystem service*") AND (landowner* OR "land manager*") AND ("United Kingdom" OR "England" OR "UK" OR "Scotland" OR "Wales" OR "Northern Ireland"). 2010-2024

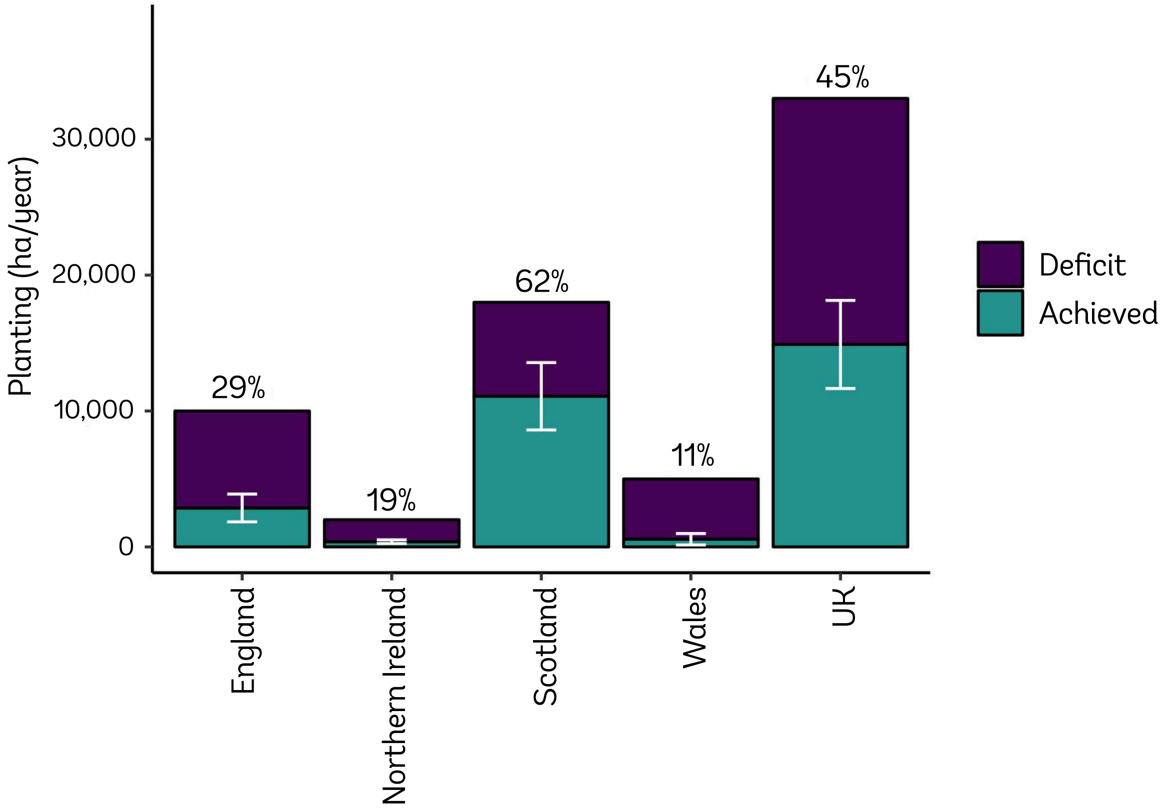
(Wood* OR forest* OR tree*) AND (Value* OR "Ecosystem service*" OR "attitud*" OR "perception*" OR "motiv*" OR "objective*") AND (landowner* OR "land manager*" OR "behav*" OR "decision") AND ("United Kingdom" OR "England" OR "UK" OR "Scotland" OR "Wales" OR "Northern Ireland")

A scoping review was undertaken to understand how values have been explored relating to woodland creation and management from 2010-2024. Due to the use of values synonymously with other terms such as attitudes, perceptions and views, additional papers were included which didn't explicitly talk about values, but which touch on these themes.

In reviewing the literature, we sought to identify: i) the main values landowners hold and which may influence woodland creation, ii) what opportunities there are to tap into these values, and iii) the gaps in our understanding which constrain our ability to do this. Here we focus on values, however, as other reviews (see Staddon et al., 2021) have noted, there are a wide range of factors which influence behaviour and whether or not these values will be realised within decision-making.

Results and discussion

Annual creation rates still aren't being met



Caption: Average area of woodland creation achieved (2020-2024) and deficit relative to CCC minimum recommendations, by country. Percentage labels show the average proportion of the recommendations achieved. Whiskers indicate standard annual deviation in woodland creation rate.

Average rates of creation have increased in every country over the past five years (2020-2024) compared to the previous reported (2016-2020). The UK as a whole achieved an average of 14,896ha per year, with this breaking down to averages of 11,084ha/year in Scotland, 2,866ha/year in England, 565ha/year in Wales and 384ha/year in Northern Ireland. However, these rates are still nowhere near the averages required to meet net zero advised by the Climate Change Committee, with the greatest deficits seen in Northern Ireland (only 19% achieved on average) and Wales (only 11% achieved on average). We are not even halfway to meeting annual targets at the UK scale (45% achieved on average).

As noted in the previous report, using extent only as a mechanism to track progress has flaws (Reid et al., 2021). Annual reporting via forestry statistics is based on very broad splits between broadleaved and coniferous woodlandat country scale. This makes it challenging to draw any conclusions about the quality of new woodland, outcomes for nature, or change at smaller scales.

The same concerns around data accuracy also still apply, as forestry statistics are predominantly based on government grant data and may miss other activity outside this. Trees outside woodlands (ToWs) also aren't included. Westaway et al. (2023) included ToWs in their analysis of UK targets, which added another 3% to estimates of woodland cover. However, this is still negligible in terms of the observed deficit relative to net zero targets.

The use of carbon related targets intended to help the UK reach its net zero targets should also be considered critically. Implying that the need for carbon sequestration via changing land use practices is the only driver for change runs the risk of pushing other objectives to the sidelines. High carbon landscapes aren't the only landscapes deserving of protection – maintaining high biodiversity within landscapes requires a rich mosaic of multiple habitats, and any net zero interventions should also support (and increase) biodiversity and bioabundance (Cole et al., 2022, carbon chapter of this report).

Understanding values

Values are a central component of a person's decision-making. They reflect people's judgments of what is important in life, as well as how we should behave. As such, they can be viewed as core motivational goals shaping people's attitudes and beliefs (Schwartz and Bilsky, 1990; Dietz, Fitzgerald and Shwom, 2005; Reser and Bentrupperbäumer, 2005; Schwartz, 2006). They are particularly relevant to decisions about whether to create woodland due to the long-term and permanent nature of this as a land use change (Iversen et al., 2022). In addition, policies and programmes to encourage woodland creation are heavily influenced by personal, cultural and institutional values (Ihemenzie et al., 2022). This means that they reflect a certain set of values, through the criteria they set for measuring outcomes, and by prioritising certain values within their goals, rules and choice of language (Chapman, Satterfield and Chan, 2019). By using values-based frameworks to understand land managers' decisions around woods and trees we can understand where values may come into conflict and align, both among land managers and with policy and available schemes and mechanisms, as well as what underpins particular attitudes and beliefs concerning woodland creation.

Values can be conceptualised in a few different ways. Transcendental values can be viewed as guiding principles that transcend specific situations, and are shaped by our exposure to societal institutions, norms, cultural traditions and laws (Schwartz and Bilsky, 1990; Karppinen and Korhonen, 2013). These values reflect the society we live in and underpin moral judgements about how we and others should behave (Reser and Bentrupperbäumer, 2005; Kenter et al., 2015). Contextual values meanwhile, are more context specific and relate to the importance or worth we ascribe to an object of value (Dietz et al., 2005). Contextual values are closely associated, yet distinct, from attitudes and preferences, which can be considered outcomes of an individual's values (Dietz, Fitzgerald and Shwom, 2005). In summary, values reflect an opinion of worth, whilst an attitude reflects positive or negative feelings towards an object or an activity, and preferences a rating of the object in question (Kenter et al., 2015). For example, we might value a woodland for its wildlife, have a negative attitude towards felling, and prefer natural regeneration as a management technique. As important underpinnings of attitudes and subsequently behaviour, it is important to understand the values of those responsible for managing land so that additional opportunities for creation and effective woodland management might be tapped into.

Typically, values relating directly to wooded habitats have been characterised as either intrinsic, where nature has value in and of itself (O'Neill, 1992), or instrumental, which relate to human uses (Kuuluvainen, Karppinen and Ovaskainen, 1996; Shanafelt et al., 2022). However, increasingly the importance of the complex relationships people have with the natural world are being recognised. This can be seen in the increased focus on the cultural ecosystem services nature provides (Fish, Church and Winter, 2016), and the idea of relational values, which describe how people relate to the natural world and to one another within the natural environment (Chan et al., 2016; Himes et al., 2024). Relational values manifest in everyday practices such as everyday activities, in the application of traditional knowledge, through language, through feeling connected to a place or social community, as part of an inheritance from former generations and through feelings of stewardship.

Exploring relational values, alongside intrinsic and instrumental values, offers an opportunity to enhance people's connections to the natural world through the other, non-environmental values they hold (Klain et al., 2017). Woodlands are a public good, so debate about their value transcends just their value to us as individuals to encompass shared social values around our relationship to nature (Kenter et al., 2015). Indeed, a review of the values associated with a single tree species, ash (Hall et al., 2021) highlights the wide range of social, cultural, economic and environmental values which may be associated with woods and trees.

It is important to note that appealing to a landowner, manager or farmer's values on their own is not enough to increase woodland cover. As noted in the funding section there are a variety of barriers which need to be overcome to involve land managers in woodland creation. Common barriers noted in the literature include a lack of knowledge (Burgess, 2017; Louah et al., 2017; de Jalon et al., 2018), support (Burgess and Rosati, 2018; Raskin and Osborn, 2019), the cost of creation (Raskin and Osborn, 2019), perceived low economic value of trees and the stressors acting on landowners such as a high workload and financial pressures (Mills et al., 2021). Understanding values is just one piece of this puzzle but can serve as a starting point to understand how these barriers affect decisions and how we might begin to overcome them.

Here we use intrinsic, instrumental and relational values to understand how land managers think about and value wooded habitats and the implications this has for uptake of schemes and mechanisms. Recognising these values and understanding which values those making decisions about woodland creation and expansion hold, may open up new potential policy approaches through which to engage landowners in woodland creation.

What values do land managers hold relating to woods and trees?

There are certain values which emerge as particularly important to landowners and managers across studies, and which have relevance and implications to how they might be engaged in woodland creation and ongoing management.

Economic values

Despite the prominence of economic arguments in woodland creation to date, economic values and objectives are often deemed to be less important by woodland owners (Lawrence and Dandy, 2014; Feliciano et al., 2017). This is aside from large-scale woodland owners managing woods for production and focused on timber as an investment (Ambrose-Oji et al., 2018; Ficko et al., 2019). This may in part be due to beliefs amongst landowners that woodlands offer little economic potential (Church and Ravenscroft, 2008; Greenslade et al., 2020; Staddon et al., 2021). Indeed 56% of respondents to the Royal Forestry Society's survey on woodland creation (2020) viewed the loss of economic value of conversion from agricultural land to woodland as a barrier. This could be addressed by improving knowledge around the economic potential of woodland, beyond traditional timber production. However, it may also be the case that landowners are simply not motivated by economic arguments when it comes to woodland. Indeed, Sutherland et al (2011) found production to not distinguish any of their land manager types, with those holding these values also prioritising community and stewardship values.

Findings on the role of incentives are also mixed. In the British Woodlands Survey, 51% of existing woodland owners stated that grant aid would encourage them to plant more woodland (Hemery et al., 2020). However, whilst grants remain appealing for investor-owned and more financially oriented landowners involved in timber production (Eves et al., 2015), there remains a significant number of owners who aren't economically minded who are missed by, or don't take up, these grants (Church and Ravenscroft, 2008; Lawrence and Dandy, 2014; Staddon et al., 2021) and so will require different approaches.

For farmers, maintaining financial security is a main driver of participation in agri-environment schemes (Sutherland et al., 2012; Coyne et al., 2021). Indeed, the diversification of a farm's business and ensuring succession to future generations were key values to emerge in discussions amongst Welsh farmers in Verfuerth et al (2023), and participants in Follett et al (2024) expressed a strong desire for market-led solutions driven by positive public perceptions of sustainably produced food. Woodland creation as a means to diversify a farm's business may fulfil this value, however the long-term change in land use when converting agricultural land to woodland may also act as a barrier, reducing the capacity to adapt to an uncertain future (Staddon et alk., 2021; Verfuerth et al., 2023). The importance of woods and trees for rural economies also appears to be an important value across studies (Bowditch et al., 2019; Cusworth and Dodsworth, 2021; Bowditch, McMorran and Smith, 2023). For example, participants labelled as 'productivists' felt afforestation should focus on productivity to improve the wealth of local communities in Scotland (Nijnik et al., 2016).

Biodiversity values

Biodiversity is often valued by landowners. Work looking into the objectives of owners has consistently found landowners rank biodiversity and conservation highly (Urquhart and Courtney, 2011; Eves et al., 2013; Ambrose-Oji, 2019; Lawrence and Dandy, 2014; Hemery et al., 2020). Wanting to increase biodiversity has also been found to be a key preference for agroforestry (de Jalon et al., 2018) and an indicator of wider agri-environment scheme participation (Mills et al., 2021). However, as research into farmers has found, the reason biodiversity is valued may vary, from its benefits for humans and food production (Sutherland et al., 2011) to the enjoyment of seeing wildlife on their land, or its inherent intrinsic value.

These different ways of valuing nature can have implications for management, which may result in tensions in terms of how to manage woodlands for biodiversity (Hague et al., 2022). Additionally, whilst stakeholders in Hague et al (2022) respected the conservation value of a woodland, they referred to this value in vague and ill-defined terms. Indeed, a lack of understanding of the ecological value of woodlands has been noted as a barrier to woodland management more broadly (Greenslade et al., 2020).

Environmental values

Other instrumental environmental values such as flood prevention have also been highlighted as important in a range of studies looking at tree planting and participation in agri-environment schemes (Mills et al., 2017, 2018, 2021; Coyne et al., 2021), with Staddon et al (2021) emphasising connecting these benefits to local communities as a key message to be used in engaging certain groups of farmers. However, when it comes to the more globalised public benefits of carbon sequestration, the picture is more mixed. Whilst the carbon sequestration value of woods has been noted as a growing motivation (RFS, 2020; Hemery et al., 2021), several studies have also found that emphasising the global importance of climate change mitigation may not resonate with farmers and other landowners (Moseley and Valatin, 2014). As Staddon et al (2021) this may shift as the impacts of climate change become more apparent at a local level.

Relational values are important to landowners and managers

The concept of relational values has gained traction over the last decade to recognise the importance of people's relationships with nature (Chan et al., 2016). Indeed, many of the values found to be important to landowners in studies, can be considered as relational values. For example, a person's social and personal identity (Warren et al., 2016), custodianship (Church and Ravenscroft, 2008; Quartuch and Beckley, 2013; Lawrence and Dandy, 2014), and landscape values (Iversen, van der V. Naomi, et al., 2022; Bowditch, McMorran and Smith, 2023; Pearson and McConnachie, 2023) have all been found to be important. Here we highlight how these relational values underpin and influence decisions about woodland creation and management.

Identity

A person's social and personal identity, relating to the significance they attach to their sense of self and membership of certain groups, can be a powerful driver of decision-making about land-use and are shaped and reinforced by social norms (Chapman et al., 2019). As a result, social norms and relationships with other land managers are key to shaping identities, with a significant body of research emphasising the importance of social networks and approval of peers (Staddon et al., 2021). These relationships, and what is deemed appropriate management are important to understand, particularly as woodland owners and managers rank other owners as the most useful source of advice (Hemery et al., 2018). Indeed, for many woodland managers, particularly those identified as traditionalists, or productivists, the role of their cultural identity as foresters has been found to be a key driver of their decision-making (Fuller and Gill, 2001; Duesberg, Dhubháin and O'Connor, 2014), reinforced by forestry's 'handshake' culture (Greenslade et al., 2021).

Most of the literature has focused on the role of self-identity in farmers' decision-making. This is often a key aspect of a farmer's culture and has been found to influence attitudes towards tree planting and woodland creation

(Thomas et al., 2015; Warren et al., 2016; Hopkins et al., 2017; Ford, Bale and Taylor, 2024). Identity can be a key value driving behaviour, in particular ideas about what a 'good' farmer is (Burton, Kuczera and Schwarz, 2008). For example, farmers may fear the loss of respect of their peers by engaging in behaviour counter to what a good farmer does (Staddon et al., 2021; Cussworth and Dudsworth, 2021). Indeed, norms and values around what crops should be grown and how land should be managed productively are culturally embedded within farming communities (Duesberg et al., 2014; Valatin et al., 2016; Staddon et al., 2021), with farmers 'reading' the landscape around them for signs of good management by their peers (Chapman et al., 2019).

This is not to say that woodland creation cannot fit within this identity, particularly as sustainability of the farm and responsibility to future generations appears to be a particularly key value held in the face of ongoing environmental change (Pearson and McConnachie, 2023; Verfuerth, Jones and Roberts, 2023; Follett et al., 2024). Research has highlighted an increased integration of sustainability within farmers' identities, owing to an increase in the environmental, economic and social capital farmers receive from scheme participation and increasing awareness of environmental issues amongst farmers' social networks (Cusworth and Dodsworth, 2021; Follett et al., 2024). This awareness could be leveraged to enable woodland creation to fit within notions of what a 'good' farmer is and does. Farmers in Coyne et al (2021), highlighted environmental benefits as a key motivator, feeling a sense of pride and personal satisfaction after planting trees, whilst Welsh farmers in Follett et al (2024), placed importance in looking ahead to future generations and the interconnectedness between environmental, social, cultural and economic aspects of trees on farms.

To facilitate this shift then, communication should aim to support and make use of local and cultural links amongst farmers (Follett et al., 2024). The social networks of farmers could also be utilised through the use of 'demonstration' farms (Hopkins et al., 2017). However, a cultural division between farming and forestry has been widely identified (Warren et al., 2016; Hopkins et al., 2017; Ford, Bale and Taylor, 2024). This divide and a lack of engagement with forestry professionals' results in a lack of knowledge and skills transfer between these two groups (Lawrence, Dandy and Urquhart, 2010). This is a particular problem as a lack of knowledge around creation and woodland management has been noted as a key barrier in numerous studies (Burgess, 2017; Louah et al., 2017; de Jalon et al., 2018). However, this divide has been suggested by Ford et al (2024) to be breaking down in England, with many participants in this study feeling that this divide was narrowing due to increasing uptake of forestry knowledge within farming networks. Helping bridge these two social networks may then help shift norms around what a 'good' farmer does and facilitate knowledge transfer between these two groups.

Woodland owners also place importance and value in having autonomy and a sense of control (Church and Ravenscroft, 2008; Urquhart, Courtney and Slee, 2012). This is an important consideration as a barrier to woodland expansion for both farmers entering forestry and existing woodland owners. Stricter regulations associated with forest governance (compared to agriculture) and specific grant regulations, along with the long-term nature of woodland as a land use change can act as a barrier (Urquhart, Courtney and Slee, 2012; Lawrence and Dandy, 2014; Valatin, Moseley and Dandy, 2016), particularly in the face of the various sources of uncertainty concerning future land use (Staddon et al., 2021; Verfuerth et al., 2023). Particularly important to woodland owners is the psychology of ownership, where their land may serve as an extension of their identity and their selves (Andabaka, Teslak and Ficko, 2021) and which owners may feel is threatened by new or changing schemes (Lawrence et al., 2010; Urquhart et al., 2012).

Landscape values

Cultural values have long played a central role for woodland owners in the UK (Urquhart, Courtney and Slee, 2010; Urquhart and Courtney, 2011; Plieninger et al., 2015). Of particular importance is the contribution of woods and trees to the character of a landscape, a value common across studies (de Jalon et al., 2018; Bowditch et al., 2019; Hague et al., 2022; Iversen et al., 2022; Pearson and McConnachie, 2023). Landscapes hold particular values and meanings for the people that live within them (Stenseke, 2018) and are shaped by both environmental and social processes. For example, Bartlett (2011) and Hague et al (2022) highlight the role that coppicing, a traditional management practice, has played in shaping UK landscapes.

It's important to note that landscape can be a 'contested term' (Langston, Ros-Tonen and Reed, 2024). Landscapes are framed by a person's values, as well as being culturally and locally informed (Iversen et al., 2022; Bowditch, McMorran and Smith, 2023). It's essential, therefore, to understand what different groups of people mean in terms of landscape value, and the role of woods and trees in this. There are long-standing cultural norms which underpin resistance to afforestation in Scotland (Hopkins et al., 2017) and the uplands in England (Reed et al., 2009; Iversen et al., 2022) due to the integral role other land-uses such as upland and moorland farming hold for local communities and traditions, as well as factors such as the long history of monoculture plantations in Scotland (Nijnik, Nijnik and Brown, 2016). The deep rootedness of these values, and their connections to identities, make these values difficult to change (Greenslade et al., 2021). As a result, landscape change has a history of being hotly contested (Reed et al., 2009) and previously, a lack of consideration of locally held values when implementing these into national policy has resulted in stakeholders having negative experiences with consultations (Iversen et al., 2022).

This contestation is evident in Iversen et al's (2022) exploration of the role of emotions and values in perspectives on upland woodland creation. In this study upland farmers held strong values concerning how the landscape should look, and its contributions to their sense of place, way of life and their local culture. Woodland creation therefore was perceived not only to change the landscape but also threaten their way of life by altering the way that the land has always been managed to earn a livelihood. The findings of Iversen et al (2022) reflect those of Duesberg et al (2014) who found farmers' decisions to engage (or not) in woodland creation were driven by intrinsic values and the belief that farming is what the land ought to be used for. However, there are situations where trees and woods do fit into ideas of how land should be used. For instance, trees within hedges were viewed as part of the landscape aesthetic for dairy farmers in the South West of England (Pearson and McConnachie, 2023). Working to develop positive ideas of how trees should be restored to landscapes and avoiding abrupt or enforced change which has created conflicts in the past (Iversen et al., 2022), may shift perceptions.

The scale of the action has a significant influence whether conflict arises. For example, in the highlands of Scotland, woodlands were viewed by estate managers as important for preserving the traditional landscape, but only as a minor component within an open landscape (Bowditch et al., 2019). In Follett et al (2024) small-scale planting in non-viable areas was linked to the long tradition of farmers as custodians of trees and the landscape, whilst larger-scale planting was linked to the loss of community and culture that has resulted from large-scale carbon-offsetting projects by external organisations. A similar concern was raised in Verfuerth, Jones and Roberts (2023) with external organisations buying up land to create woodland being seen as threatening to the way of life of local Welsh communities. Encouraging small-scale planting may therefore be more likely to align with the values of those not engaging with existing schemes, particularly as a lack of options for this scale of planting and for agroforestry has been noted as a key barrier (Ford et al., 2024).

Cultural and historical values and the role of stories

Stakeholders in Hague et al (2022) drew on stories of cultural or historical land use to argue for more material use of the wood, whilst also recognising this could impact what makes the wood so special. Narratives of this historical use highlight how humans have shaped the landscape for hundreds of years, providing the habitat wildlife relies on. Recognition of the active role of people in producing and shaping the values that habitats such as woods and trees provide has grown in recent years (Chan et al., 2016; Fischer and Eastwood, 2016). However, there has been tension between this historic cultural value and traditional biodiversity conservation, which has been grounded in a feeling that managing for conservation meant removing human activity from the woodland:

Hague et al (2022) concluded that stories involving humans were more powerful in driving engagement than stories of nature on its own, and the discourses and narratives surrounding landscapes and governance issues have also been explored elsewhere, with the long-standing influence of humans present in many narratives around wooded habitats (Walsh, 2020; Langston, Ros-Tonen and Reed, 2024; Schaal-Lagodzinski et al., 2024). This highlights how cultural values, particularly around the history of human use of woods, can cut across other values, from aesthetic or economic values to their importance for rural livelihoods and local communities. For instance, Bowditch et al (2019) highlight how estate managers in the highlands feel a responsibility to maintain cultural landscapes owing to their importance for the public and local traditions such as deer stalking, whilst participants in Hague et al (2022) advocated for increased future use of woods to get people back into the landscape. Community concerns such as employment and benefits to rural life were also given importance by land-managers in Scotland (Sutherland et al., 2011).

A particular area in which the value of tradition and history of managing the land resonates is in the value given to being a custodian of the landscape.

Indeed, custodianship is a value shared by many woodland owners (Church and Ravenscroft, 2008; Sutherland et al., 2011; Quartuch and Beckley, 2013; Lawrence and Dandy, 2014; Greenslade et al., 2020; Bowditch, McMorran and Smith, 2023) and is often driven by a sense of moral responsibility to take care of the environment (Mills et al., 2018, 2021). In Follett et al (2024) there was a shared feeling of pride in farmers' roles as custodians of the land, with coppicing and hedgerow trimming highlighted as examples of traditional environmental guardianship that fulfil this role. However, custodianship and similar concepts such as stewardship can have multiple meanings, not all of which can be assumed to be beneficial for nature (West et al., 2018). It's therefore essential to understand what this custodianship means to different landowners and farmers, so that woodland creation can have the best possible outcomes for people and nature.

Using understanding of values to influence woodland creation

To fully understand the values held by landowners and managers towards woods and trees, there is also a need to consider these alongside the other values they hold, and how these might link to their receptiveness to engage in woodland creation. For instance, in Nijnik et al (2016), all groups supported the expansion of woodlands in Scotland, despite a diversity of attitudes and priorities for what this might look like exactly in terms of objectives and resulting woodland composition. The following studies highlight opportunities to tap into multiple values when engaging landowners and managers in woodland creation.

Agroforestry involves incorporating trees into farming systems and represents a key opportunity for increasing woodland and tree cover across the UK, whilst also increasing the benefits agricultural land provides (i.e. increasing soil productivity, carbon sequestration, water management and livestock welfare) (Felton et al., 2023). However, uptake has remained low (den Herder et al., 2017), and there is a need to better understand what values might underpin farmers' engagement in agroforestry. Dairy farmers have been found to be amongst the most willing to engage with creation, despite this group, at 2% of the total farmed area, in England, having the lowest level of existing woodland (Pearson & McConachie, 2023). Dairy farmers interviewed found multiple values important, with livestock health emerging as a key value through which tree cover could be increased. This value was multifaceted, relating to the value of the farm as a business, for food production and as being central to a farmer's identity and way of life. As such, this represents an opportunity to appeal to multiple values simultaneously. The authors concluded by emphasising how woodland creation can align with dairy farming, suggesting this would be more effective than suggesting alternative livelihoods (i.e. woodfuel). However, to do so requires overcoming some farmers' negative perceptions of the impact of trees on livestock health.

Verfuerth, Jones and Roberts (2023) found Welsh farmers placed importance in similar values. Two main themes arose from their participatory backcasting approach: the important role farmers have and will have as food producers for Wales, and the importance of farms to local communities, especially Welsh-speaking communities. These values were linked to their identity as farmers, a desire to feel valued for their contributions and to have agency and control over food production. Whilst this sometimes conflicted with woodland creation, farmers also saw creation as having potential to support diversification of farm income, particularly when looking forwards to the future. The need to diversify to increase the viability of farming for future generations was raised as a key concern. However, there were also concerns, owing to the long-term nature of woodland, that converting areas of farms to woodland might also reduce their ability to adapt in the future. In addition, larger-scale tree planting was seen to be in opposition to preserving the culture and tradition of farming within Welsh-speaking communities.

Meanwhile, Bowditch, McMorran and Smith (2023) used walking interviews to gain insights into woodland culture of private sporting estates in Scotland, finding production values to be important to all but two estates when making decisions about trees. Whilst an important driver, they also found that managers held additional values concerning the aesthetic value, biodiversity value and contributions to the landscape value of non-commercial woodland, which led to them questioning the place of production. Deer stalking was viewed as a local tradition that estate owners had a responsibility to the local community to preserve (Bowditch et al., 2019; Bowditch, McMorran and Smith, 2023). In addition, native woodlands were viewed positively due to their contribution to the landscape aesthetic and for requiring minimal management input as well as supporting the estates' sporting interests. The importance of woodlands for deer stalking led to positive views of natural colonisation. This method of creation was viewed by most owners as an acceptable and favoured option for shelterbelts as they expected a more natural woodland structure to emerge compared to planting, providing varied habitat and shelter for deer and contributing to better quality stalking. The lack of commercial opportunity for wood products, but a perceived value as habitat for deer stalking, highlights how economic values can be utilised within this context to encourage the creation of native woodland.

Finding shared values

For land use change to occur at the scale needed to meet ambitious and urgent targets, there is a need to understand where the values of landowners and managers might align to develop consensus and foster collaboration. Values can be used to bring together like-minded landowners and bridge gaps, enabling people to feel like they can better align their practices with their values. To do so requires facilitating relationship building by finding common ground (Bowditch, McMorran and Smith, 2023; McConnachie, Pearson and Spencer, 2023). For instance, Burton et al (2019), in exploring the shared visions of stakeholders including land-managers, found varying degrees of overlap around land sparing, conservation, utility and land sharing, with common ground centring around value for carbon sequestration, water and biodiversity benefits. To utilise these shared values however, required acknowledging the differences in how these values were perceived and how they should be translated into management actions.

Implications for policy

Governance should be tailored to local contexts

A frequent cause of conflict for stakeholders relating to woodland creation is a lack of feeling heard, or previous negative experiences with consultations (Iversen et al., 2022). There is a need to tailor national targets and policies, schemes and mechanisms to local contexts which account for local cultural values. This could help to address the issue of mismatched scales, where national policy fails to align with local values (Raum, 2017; Bowditch, McMorran and Smith, 2023). Achieving this will require developing channels that facilitate dialogue between local and national level stakeholders and provide a forum for local input.

Facilitating dialogue links to research which has identified a comprehensive land use strategy as the most important policy to prioritise in terms of delivering land use transitions (Ford and Taylor, 2024). Although developed for Scotland, this is lacking for other devolved nations, and for the UK as a whole.

Should a land use strategy (or strategies) be developed, it will be essential to bear in mind the scale at which any spatial planning is carried out, as well as the extent to which some objectives are prioritised above others. It has been highlighted that there are multiple contradictions in differing land use pathways to net zero (Cole et al. 2022). To mitigate these, we need to make use of transdisciplinary approaches where there is increased local and devolved decision making, and where no single land use solution is promoted above all others (Cole et al., 2022).

Tap into values

A wider recognition of the multiple values which underpin attitudes and beliefs about woodland creation and restoration will enable more effective collaboration and could increase engagement with creation using tailored campaigns (Moseley and Valatin, 2014).

For example:

- Tree planting campaigns targeted at farmers, could emphasise the benefits of trees to the farm business and its sustainability, to align tree planting with the 'good' farmer identity (Coyne et al., 2021; Follett et al., 2024). For instance, through linking agroforestry schemes to positive public perceptions of sustainably managed food (Follett et al., 2024).
- These campaigns could also tap into the importance placed on connections to local communities (Verfuerth, Jones and Roberts, 2023; Iversen et al., 2022; Moseley and Valatin, 2014). For example, by highlighting benefits to local communities such as reducing flood risk, soil erosion, increasing biodiversity and contributing to a sense of place.
- The historical role of woods and trees in the landscape (Pearson and McConachie, 2023) could also be utilised in these campaigns.
- There is a significant body of research emphasising the importance of social networks and approval of peers (Staddon et al., 2021). Communication should aim to support and make use of local and cultural links amongst farmers. For example, through the use of 'demonstration' farms (Hopkins et al., 2017).
- Farmers and other landowners may feel that creation can conflict with their other priorities and values (i.e. reducing agricultural pollution, increasing water storage) (Verfuerth, Jones and Roberts, 2023). Communication should recognise these multiple pressures and could emphasise how trees can help address these challenges.
- Campaigns and schemes for specific groups of land-managers (i.e. groups

with historically low levels of woodland cover) could be tailored to the particular values which resonate for these groups.

Evidence gaps and future research needs

The range of methods used to understand decision-making around woodland creation and management has expanded in recent years. Recent studies highlight how a wider range of methods can be used to deepen our understanding and uncover ways in which a wider range of land managers can be engaged to meet national creation targets. These methods will be particularly important for addressing remaining evidence gaps and improving the uptake of woodland creation amongst diverse groups of land managers. For example:

- As argued in Staddon et al's (2021) review of creation on farms, the usefulness of additional quantitative typologies and segmentation models is arguably limited. Indeed, there is a need for more mixed-methods approaches to understand values around woodland creation, expansion and management. In a review of European forest owner typologies, Ficko et al (2019) noted that only 10% used mixed methods. More qualitative and mixed-methods approaches will enable us to deepen our understanding of what the values that are held for woods and trees, and how we might facilitate and encourage their enactment.
- There is a need to co-design policy so that interventions are fit for purpose and more likely to be taken up by intended end users. There are a range of participatory approaches which could be used (Burt, Mackay and Mendibil, 2021; Verfuerth, Jones and Roberts, 2023). The choice of these methods should be tailored to the decision-making context and specific research questions, as well as follow best practice around the use of participatory methods (Maund et al., 2022).
- Exploring the differences between those participating and not participating in schemes has been noted as a gap by Coyne et al (2021). This is likely to be difficult to address due to the difficulty in reaching non-participants/ those not interested in creation (Coyne et al., 2021; RFS, 2020), so may require novel participant recruitment methods and avenues through which to engage in research.
- There is a need to explore how woods and trees are valued and considered in comparison to other land uses, particularly other sustainable land management actions available to those managing land (Follett et al. 2024). This will enable us to better understand how woods and trees may fit within multifunctional landscapes.
- A values framework could also aide in understanding willingness to engage in different creation techniques such as natural colonisation, which might conflict with values around 'tidy' landscapes (Chapman et al., 2019; Nassaeur et al., 2009).
- Emerging stakeholders in woodland creation such as corporations and large landholders (Staddon et al., 2021) may hold different values for nature which motivate their participation in schemes. There is a need to further explore their values to maximise the potential of this group of stakeholders.

- There is a need to understand how common ground can be found between buyers and sellers in schemes such as the Woodland Carbon Code, particularly as mechanisms such as green finance develop further. Aligning the values of buyers and sellers has been found to be a key part of this process (Koronka et al., 2022). Therefore, the potential of these schemes requires understanding how to bring together these two groups.

Values in a changing world: applying a value lens to understand the challenges facing woods and trees.

The literature explored here has covered the range of values that landowners and managers may hold for woodland and trees primarily in the context of creation. However, exploring values as fundamental drivers of behaviour can also offer insight into other issues facing the UK's woods and trees. For example, Young et al (2018) explored how values, knowledges and practice linked to understandings of forest resilience. Additionally, Dandy et al (2012) explored beliefs around deer management, grouping these into five categories: naturalness, overabundance, impacts, effectiveness and animal welfare. Hall et al (2021) meanwhile explored values in connection to ash trees, exploring how this influences how the impacts of ash dieback are managed and highlighting the range of values that may be attributed to a single tree species.

Exploring the values held for a wider range of wooded habitats, such as ancient woodland for example, may offer insight into how people conceive this habitat, and how they might seek to address the threats it faces. Values may also play a role in decisions about tree provenance, and in the management of species such as deer and grey squirrels. Applying a values lens to these problems could unlock new insights into decision-making and offer up new policy solutions.

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Woodland management

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The benefits of woods and trees are increasingly known (see earlier chapters of this report), and policy has responded to this with a focus in recent years on new woodland creation, with tree cover expansion targets set.

We know that diverse habitats are more robust and stable than diminished ones (Tilman et al., 2014) and that the integrity of an ecosystem is fundamental to the maintenance of functioning, resilient woodland systems. Because of this, simply creating new woodlands cannot increase resilience in the face of unpredictable threats such as climate change. This is because it can take decades before these new woodlands are able to deliver benefits. As well as more woodland creation, there also needs to be appropriate management of existing woodlands to help protect their functioning. Whilst important, woodland creation on its own will not deliver the multitude of benefits our woods and trees provide. We know that currently the majority of Great Britain's woodlands are not in good ecological condition (no data exists for Northern Ireland) (Reid et al., 2021); their structure and composition has become more simplistic over time; and that they face a barrage of threats that threaten their condition (condition chapter this report). To move our woodlands into better condition and to maintain this, appropriate management is required.

In 1944 at Lady Park Woods in Monmouthshire, a non-intervention approach was established to understand what would happen to an ancient woodland left without intervention. The woodland has seen a reduction in species diversity over this time and the reserve is rated as 'unfavourable, declining' for condition (Peterken and Mountford 2017). This is thought to be due to excessive levels of deer grazing and the lack of intervention to reduce these deer impacts. In woodlands adjacent to the site, fencing has excluded grazing which has led to excessive bramble and bracken. It was also shown that in humid woodlands with acid soils and invasive ***Rhododendron ponticum*** present, no intervention led to increased shading and diversity loss. These examples show how appropriate management enables diverse and functioning woodlands (Kerr and Mason 2019).

In addition to the environmental and ecosystem service benefits of managing woodland, there are also economic and social benefits (Kerr 2020). The Independent Panel on Forestry (2012) stated that managed woodlands can provide better landscapes and have greater amenity value and can confer significant health and wellbeing value to people.

The sustainable production of timber and wood fuel can provide significant economic benefits. The UK Confederation of Forest Industries estimated that England and Scotland imported 32,000 tons of firewood in the first nine months of 2017, but that this could be produced by just 8,000ha of managed broadleaved woodland (Harris 2019). This could reduce the reliance on imported wood products as well as boosting economies. An additional benefit of reducing the reliance on imported timber is lowering the biosecurity risk. Imports have been the source of Dutch elm disease, the great spruce bark beetle and Asian longhorn beetle (Brasier 2008). The economic cost of just six of these pests has been estimated to be around £919 million (tree health section, this report).

What do we mean by management?

The definition of management varies. Some examples include:

Chartered Institute of Ecology and Environmental Management (CIEEM) and The Institue of Chartered Foresters (ICF) (2023) define management as '*the active oversight of a whole site to ensure the woodland and any activities deliver the intended benefits in accordance with UK Forestry Standard (UKFS)*'. Intervention is defined as '*a specific activity or group of activities intended to meet a woodland management objective*'.

Forestry England defines woodlands as 'sustainably managed' if '*the woodland is managed to the UK Forestry Standard that has a woodland management plan, or for which they have provided a grant or a felling licence in the last 15 years. It also includes all woodland in the nation's forests managed by Forestry England, and all woodland on the Defence Infrastructure Organisation training areas. It is recognised that other woodland might be considered as managed as well.*'

Wildlife and Countryside Link (2024) defines woodland management (for England) as '*the development and delivery of a woodland management plan to deliver objectives, which might include, for example, timber production, public access, carbon capture, and crucially, should include improvement of woodland ecological condition and biodiversity conservation. Management can include minimal intervention where it best delivers set objectives. Indiscriminate felling in woodland cannot be considered as management*'.

The more holistic definitions from CIEEM and Link allow the nuance that management will vary depending on the age of the woodland (ancient, secondary, new), type of wood (e.g. temperate rainforest) and what the management objectives of the woodland are (e.g. timber production, recreation, biodiversity conservation, carbon capture). To assist nature recovery, woodlands should be managed for diversity and good condition.

How much of our woodlands are currently managed?

One of the reasons for woods falling into unfavourable or intermediate condition (Forest Research 2020) is the historic management of woodlands. Last century, policies encouraging clearance of low-yielding native woodlands at scale to 'improve' them with higher yielding exotics was responsible for much ecological degradation of ancient woodland sites. More recent factors such as herbivore damage and continued undermanagement have also contributed to the number of stands in unfavourable or intermediate condition. Current inappropriate management, or lack of management, also plays a significant role.

The number of UK woodlands under management is hard to ascertain due to a lack of data. Proxies can be used but they tend to be based on the less holistic definitions of management and so understanding the proportion of woodlands that are under *appropriate* management is almost impossible to ascertain.

Figures for certified woodland areas are often used as an indicator of sustainable forest management. Certified woodland in the UK has been independently audited against the UK Woodland Assurance Standard (UKWAS) since the late 1990s. In 2024, the total area of certified woodland in the UK was 1.44 million hectares (44% of the total UK woodland area). This is 60% of woodland area in Scotland, 56% in Northern Ireland, 48% in Wales and 23% in England (Forest Research 2024). These figures relate to all woodland within the UK, and the majority of certified woodlands may be productive conifer forests. Woodland that is not certified, may also be managed sustainably. The current methods for measuring managed woodland have been questioned, as the threshold for being classed as 'managed' does not make any commentary on the appropriateness of the management. In addition, there is no way to track whether management goes ahead; management plans may be approved but not implemented. We require better measurements for measuring appropriately managed woodlands in order to gain a clearer understanding of the current state.

In addition to these official statistics, the Bunce report (Smart et al., 2024) shows there was a decrease in interventionist management across surveyed sites over 50 years: while there was evidence of historic coppicing and/or tree clearing in around 60% of the plots, only 10% of plots showed evidence of recent canopy gap creation in the 2022 survey compared to 18% of plots in the earlier surveys. It's important to note that 'management' in the Bunce report refers to intervention to create canopy gaps, rather than the more holistic definitions of management.

The Royal Forestry Society (2019) has recently reported that most unmanaged woodland is broadleaved and in private ownership and estimated that the area of unmanaged woodland that could feasibly be brought back into pro-active management (physically and economically) is up to 200Kha in England and 53Kha in Wales. This could generate up to £20million worth of home-grown timber and wood fuel a year and support 240 rural jobs in the supply chain.

What can be done to improve the condition of our woodlands?

In this section we explore how appropriate management and interventions can help to improve the conditions of our woodlands. Providing a comprehensive set of advised management actions is beyond the scope of the report, and the aim is to demonstrate the current knowledge base available. A Royal Forestry Society report from 2019 provides links to resources to help land managers on the different aspects of forest management connected to resilience, from climate change impacts and adaptation to surveying and maintenance of the soil resource Resources-for-Managing-Woodland-Resilience.pdf

Management interventions to increase forest resilience can take many forms. Managers need to consider (1) whether management is needed at all and (2) how management mimics, or is a substitute for, natural processes.

The good news is we already know many of the solutions and interventions to help improve woodland condition (Table 1). The most appropriate or priority interventions will vary based on current threats, woodland type, current condition status and location, but will also be driven largely by site objectives (Harmer et al., 2010). The key to making appropriate decisions around management necessitates a connection to and knowing the woodland, as well as good monitoring over time. Management to increase woodland resilience takes a combination of reducing threats and increasing a diversity of structure and composition within the wood (see condition chapter this report).

Indeed, threat reduction may be particularly important when it comes to reducing the impact of deer. Deer play an important role in woodlands but deer numbers (native and non-native) are now too high for our woodlands (and other semi-natural habitats) as deer browsing has a major impact on woodland vegetation (Eichhorn et al., 2017; Fuller and Gill, 2001, and deer chapter this report). Deer browsing in high numbers dramatically alters woodland species composition and structure by removing understorey vegetation and inhibiting natural regeneration. Overall, biodiversity can be depleted and ecosystem functioning weakened (Spake et al., 2020).

Other threats that require management include reducing invasive non-native species such as *Rhododendron ponticum* (which is unaffected by deer numbers or pressures from other grazing animals) and grey squirrels which are damaging to biodiversity and ecosystem functioning. These non-native species dominate their environment to the detriment of other native species and can cause significant environmental, social and economic damage (Defra, 2015; grey squirrel and ancient woodland restoration chapters of this report).

Pests and diseases pose a great threat to UK woodlands (see tree health chapter this report). Taking steps to reduce the risk of introducing or spreading pests and diseases must go alongside management that minimises their impact once they have arrived and established.

There is also an increased need to proactively plan to respond to the changing climate. Climate change brings uncertainties that make building resilience and reducing the stresses on woodland ecosystems crucial to give them the best chance of survival (for example, see extreme weather events and provenance sections of this report.

In the ecological condition chapter of this report, we explored the data surrounding the individual metrics of condition. Table 1 shows currently used management interventions for each of these metrics that could be used by woodland managers.

Table 1: Known interventions that can be taken for each of the condition attributes.

Condition metric	Possible interventions
Vegetation	Gap creation to allow light to reach the understorey, conservation grazing to prevent domination by one or a few species.
Tree health	Biosecurity measures to prevent new pests and disease entering the UK. Monitoring of current pests and disease to understand presence and spread. Cleaning vehicles, equipment and personal protective equipment (PPE) between sites. Link to tree health section

Condition metric	Possible interventions
Invasive species	Grey squirrel control such as immunocontraception and culling. Invasive plant removal and control. UK-wide monitoring of invasive non-native species to inform eradication strategies. Link to grey squirrel and rhody sections
Herbivore damage	Appropriate levels of culling to reduce deer numbers. Fencing to keep deer out of stands and allow for regeneration. Link to deer section
Regeneration within stands	Gap creation to allow light to reach understorey. Ground preparation and disturbance.
Regeneration around stands	No sharp boundaries - creation of ecotones Buffers from agriculture and other intensive land uses.
Deadwood volume	Leave standing and fallen deadwood. Creating wood decay.
Veteran trees	Protection and management of future veterans by keeping them open grown, allowing adequate light and preventing competition from younger trees. Microhabitat creation. Link to AVT section
Vertical structure	Selective thinning, continuous cover forestry, coppicing (where evidence of historic coppicing) and pollarding to increase diversity of layers present in the canopy.
Age distribution of tree species	Selective thinning to increase age classes. Active planting/seeding.
Number of native tree and/or shrub species	Introduction of appropriate native species, creating open habitat to facilitate natural colonisation, and potential planting where appropriate, of absent species.
Nativeness of occupancy	Preferential thinning to favour native species.
Proportion of open space	Selective thinning and variable density thinning; creation and maintenance of open spaces, rides, glades to improve the horizontal structure; grazing with large herbivores.
Proportion of woodland / favourable habitat	Buffers from unfavourable habitat.
Size of woodland parcel	Expansion of existing woodland, and creation of new woodland, targeted to promote the connectivity of existing habitat.

It is important to keep in mind that some condition metrics may require significant time to improve (e.g. an increase in deadwood, veteran tree numbers) and so noticeable improvement may not be seen for some years or even decades (Watts et al., 2020).

What about newly created woodlands?

Some of these features (veteran trees, deadwood, age distribution of species) may take many decades to develop, making them difficult to achieve in newly created woods. Because of this, new and young woods are unlikely to score 'good' on condition assessments for certain attributes. However, well-designed woodlands, even when new can still score well on condition, and as with all woodlands should be monitored to allow them, over time, to move into good condition categories. They can be planted with mixtures, open space designed in, and threats managed. There are also many interventions currently used to mimic mature woodland attributes that could be appropriate to enhance the condition of these younger woodlands, such as provision of artificial structures for cavity-dependent species, promoting deadwood habitat formation, microhabitat formation, tree species reintroductions, population reinforcement and assisted colonisation, increasing structural complexity, and grazing with a small number of large herbivores such as cattle (reviewed in Hornigold, 2022).

Why are UK woodlands not being managed?

There is a good understanding of what we can do to improve the condition of our woodlands, and the urgent need, so what is stopping us getting there? Most UK woodlands are under private ownership and so incentives and support are needed to work for private landowners.

However, support for appropriate management appears to be lacking in some major ways.

Economics

Appropriate and sustained management of woods can genuinely benefit from long-term financial support. A review of current funding opportunities across the UK (funding section this report) highlights that there is limited financial support for management. For example, there is no government funding in Northern Ireland for woodland management outside the establishment phase for new woodland. The focus of funding is mainly targeted at woodland creation rather than looking after and improving our current woodlands. Specifically, there isn't much support for collaboration which is particularly important for factors such as deer management that is much more effective when undertaken at a landscape scale rather than by individual land managers. Support for trees outside woods and ancient and veteran trees is also lacking. Given their huge biodiversity value, increased funding for their management is vital.

The Royal Forestry Society (2019) reported that government grants have become increasingly unattractive, restrictive and unfavourable to support sustainable woodland management. The Natural Resources Wales SoNaRR reports have shown there has also been a downward trend in woodlands under management in Wales which is primarily due to the cessation of Welsh

Government's Glastir Woodland Management scheme in 2016.

Timescales of funding also need to work for landowners as woodland management and improvement in condition as a result are not possible in the short-term (CIEEM & ICF, 2023). Given the current state of flux that many grant schemes are in, this can be difficult for woodland managers to maintain management in the long-term.

In addition to a shortage of funding, the market for wood products may also act as a disincentive for some woodland owners; there needs to be an economy that provides a market for wood products. Many traditionally managed broadleaved woodlands ceased being managed in the last century due to changes in local and regional economies. A Forestry Commission report from 2019 showed that whilst timber and woodfuel prices were higher than they had been for many years, the operating costs were so high that the returns for broadleaved woodlands were modest, especially compared to other land uses. This could lead to land managers viewing woodland management as a low priority compared to other income generating land-uses.

Skills gaps

It is increasingly being acknowledged that there is a skills crisis across the environmental professions and an urgent need to expand and diversify the workforce (skills section this report). There is a shortage in professional foresters, ecologists and the contractors needed to carry out the woodland management work. There is also a lack of providers to train the next generation (CIEEM and ICF, 2023). A report commissioned by the Royal Forestry Society in 2017 identified a shortage of forestry contractors able to undertake appropriate management interventions. In addition, they reported that the hardwood supply chain has a high proportion of sole traders who have a limited capacity to respond to growing demand for harvesting and processing operations. This results in land managers struggling to secure contractors, particularly for small and more complex operations. Some of the interventions which may help to improve ecological condition of woodlands, such as microhabitat creation and sensitive thinning require specialist skills; reporting from the Green Recovery Challenge Fund (2021) showed that contractors able to carry out this sensitive work are low in number.

Advice

Often landowners may wish to carry out management but do not know what or how. A study in England (Eves et al., 2015) found that 37% of the woodland owners could be defined as "aspiring managers". These are managers who are newer to woodland ownership but require support and guidance on getting started. Access to advice from professionals and thorough guidance on woodland management is essential. Some of the grants including some support for advice exist, but more support in this area could help.

Disappearance of the woodsman

There has been a shift in who is working in and using woodlands, with a recognised loss of wood culture, and the skills and knowledge associated with this (outside the professional workforce). Many woodland owners are not professional foresters and may not engage with traditional forestry practices. Associated with skills and advice, this lack of knowledge about the sustainable management of woodlands is a barrier. Research also shows that knowledge and attitudes of landowners play a significant part in what, if any, management may be undertaken, and greater understanding of these drivers are important (see creation section of this report).

Conclusions

The multitude of benefits that can be gained from managing woodlands appropriately requires support and incentives that allow landowners to realise and achieve these benefits. It is well accepted that there are minimal risks to biodiversity from the recommencement of management as many studies show that broadleaved woodlands are at their most diverse when canopy cover is well below 100% (Harmer et al., 2010; Peterken and Mountford 2017).

The ecological, economic and social benefits of bringing woodlands back into appropriate management cannot be ignored. The current condition of our woodlands shows the urgency of this; woodlands in planned management will be making the greatest contribution to ecosystem service provision and increasing resilience to change.

Implementing actions to increase forest resilience is challenging because of the diverse and complex set of inter-related issues that currently threaten woodlands. However, action is urgently required (Tew et al., 2021). Time invested now into bringing our woodlands into good condition will pay dividends in future. Forward-thinking decisions will enable woodlands to survive and hopefully thrive under future conditions.

Evidence gaps

There is little information about how effective different management interventions are to achieve a whole range of objectives. Tracking and improving this will offer great gains for biodiversity and carbon storage.

Better data on the type of management being undertaken across UK woodlands would allow us to have a greater understanding of how many woodlands are in management and in what type of management.

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Ancient woodland restoration

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Ancient woodlands, by definition, have developed over centuries and are known to be one of the UK's richest and most complex terrestrial habitats. These woodlands have a long, uninterrupted presence (continuity) and so are often associated with high biodiversity. Ancient woodlands are made up of complex soil systems, decaying wood that provides homes to fungi and invertebrates, epiphytic groups, ground flora, shrub layers, unique rare and threatened wildlife as well as the trees themselves. For many species of animal and plant, ancient woodland sites provide the sole habitat, and for many others, conditions on these sites are much more suitable than those on other sites. Ancient woodlands are referred to as being irreplaceable. This is because it would take centuries to replicate the ancient woodland communities and the services they provide; they cannot simply be created.

In addition, ancient woodlands store and sequester carbon. As reported in *State of the UK's Woods and Trees 2021*, whilst ancient woodland makes up 25% of all UK woodland, it holds 37% of all the carbon stored in woods and trees. Ancient and long-established woodland stores an estimated 77 million tonnes of carbon in the biomass of living trees. That volume is set to increase over the next century to c. 155 million tonnes, due to the current age composition of ancient woodlands and the potential growth of younger trees (Reid et al., 2021).

There has been debate as to whether carbon dynamics in ancient woodlands approaches an equilibrium as they mature and growth rates slow, with losses from decomposition and respiration balancing carbon assimilation via photosynthesis. However, as reported in the carbon section of this report, some studies have begun to show that when you take into account the net exchange of carbon, ancient woodlands can still sequester around 1.2 tonnes of carbon per hectare, per year (1.2 t C ha^{-1} yr^{-1} in an ancient semi-natural woodland (Thomas et al., 2011) , making them net carbon sinks and further increasing their importance as a large carbon store.

The role of ancient woodlands as a net carbon sink is likely determined by historical management and disturbance over previous decades which has facilitated the regeneration and recruitment of new, younger trees with considerable growth potential. We lack data on the impact of ecological condition and management interventions on existing ancient woodlands' carbon stores, as well as future sequestration potential. Increased threats from pests and diseases, impacts of infrastructure development and pollution, and a lack of appropriate management may threaten the long-term stability of existing carbon stores and reduce their potential to continue to act as significant carbon sinks.

Ancient woodlands also hold significant cultural value, rich with historical features and artefacts. Historic woodland management for industry and crafts have had an impact on the function and structure of these woodlands, long after this management has ended. Ancient woodlands also provide rich spaces to connect with nature, benefiting both physical and mental wellbeing (see access and wellbeing chapter of this report).

Ancient woodlands play a key role in the resilience of our landscapes. They provide complex, mosaic habitats for species, act as stable carbon pools and provide quality access to green space. But they are also under a barrage of threats that affect their ability to support these services. Because of the long time these woodlands have taken to develop, when they are gone, they are gone for good. Planting and creation cannot replace these unique habitats. This means they are particularly vulnerable to change and need to be protected and restored.

Why do ancient woodlands need restoring?

As shown in chapter one of this report, ancient woodland is rare, making up around 2.5% of the total land cover of the UK. This is because ancient woodlands have faced a barrage of historic and current threats. Development, pests and diseases, invasive species and historic planting of monocultures over the sites (plantations on ancient woodland sites or PAWS), climate change and surrounding intensive land use (Schulte to Bühne & Pettorelli 2023). In addition to fragmenting them and reducing their extent, these threats have also affected their condition and subsequent resilience and their ability to provide services and habitats are weakened.

Restoration of ancient woodland is vital and urgent. However, proper restoration takes time; some processes can take 20 years, meaning that restoration needs to begin as soon as possible. Priorities for restoration include the conversion of PAWS to native woodland and the removal of rhododendron (Woodland Trust 2020).

Plantations on ancient woodland sites

What are PAWS?

Many commercial forestry plantations were established in the 20th century to reduce our reliance on imported timber, and this is still the case today. As a nation we need commercial forestry and a sustainable timber supply and the year-on-year increase in the amount of all woodland, including commercial plantations, is encouraging (see creation chapter this report).

PAWS are ancient woodlands where the native species have been partially or wholly replaced with one or multiple non-native species (usually conifers). Around 40% of our ancient woods have been converted to PAWS.

PAWS are often heavily degraded and have multiple issues that threaten their health and resilience, for example:

- Loss of ancient and veteran trees and the associated woodland flora from overshading and competition from maturing and mature non-native species.
- Regeneration of non-native species out-competing native regeneration.
- Invasive non-native species (INNS) such as rhododendron outcompeting and shading out native tree species regeneration and woodland flora.
- High herbivore impacts from deer which can lead to reduction or total loss of native regeneration and woodland flora.

Despite this, PAWS usually contain remnant ancient woodland features such as ancient woodland plants, deadwood and veteran trees which provide the building blocks to enable restoration.

Why is PAWS restoration important and what needs to be done?

Converting PAWS back into native woodland will help restore the ecological functioning of these habitats and allow biodiversity to increase.

The need to restore is more urgent than ever as we are at a unique point in time. Most PAWS are now at, or beyond, the age when they will be felled and their future decided. This future could be clear-felling and replacing with another non-native conifer plantation, or, beginning the process of restoration to help improve their condition and ecological functioning. Under the first scenario, (the most economic option) we could see a widespread repeat of the past, and further degradation and loss of these habitats for future generations. In England, felling licence provisions provide a loophole whereby actions which are legal, but harmful to nature can fall under the sustainable woodland management definition. Currently, the Government cannot grant felling licences with conditions in place to ensure felling activity is delivered in line with nature's needs. Under current conditions, an ancient woodland replanted with damaging conifers would currently count towards the Forestry Commission's sustainable management target. This of course, also does not incentivise PAWS restoration.

Restoration of PAWS does not have an official 'endpoint' where they become reclassified as ancient semi-natural woodland, although in Wales they become recorded as 'restored PAWS' (RAWS). To bring a PAWS woodland into a good condition and bring about true restoration requires a concerted effort over many years and depending on the deterioration of the most damaged sites, restoration is a process that requires decades (Woodland Trust 2020).

Where are PAWS and who owns them?

UK-wide data on conifer PAWS is available from the four country ancient woodland inventories which are currently being updated. These updated inventories will give us a more accurate understanding of PAWS extent and distribution across the UK (see extent chapter this report).

A total of 20% of all ancient woodland is owned by the national forestry agencies across the UK and is predominantly made up of plantations on ancient woodland sites (c. 91,000ha; 34% of all PAWS). The other 80% of ancient woodland is owned by private landowners and charitable organisations and other public bodies such as the Crown Estate. Of this, one third is plantations on ancient woodland sites (c. 177,000ha; 66% of all PAWS) (Reid et al., 2021).

How much PAWS is currently in restoration?

Restoration of degraded ancient woodland has been supported by government policy since the Broadleaves Policy (1985). Across the four countries of the UK there is differing policy and support for PAWS restoration. However, it is very difficult to make a clear assessment of the restoration progress on the 66% of PAWS not owned by the national forest agencies as there is no complete data available. Here we provide data from the Woodland Trust and forest agencies.

In the *State of the UK's Woods and Trees 2021* it was reported that since 2015, the Woodland Trust has assessed the condition of 21,547ha (c. 7.2%)

of PAWS on privately owned land, of which 1,636ha was classed as critical, 17,399ha threatened, and 2,512ha secure.

As a result of the Woodland Trust's work since 2008, nearly 31,000ha of ancient woodland was committed to restoration, with areas increasing every year. This includes land for which management actions have been agreed and the land manager has confirmed that they intend to carry out the work, but the work may not have started yet – with lack of funding often quoted as a reason (see funding section of this report). Since *State of the UK's Woods and Trees 2021*, there have been 4,020ha committed and 2447ha of PAWS in active restoration.

The Woodland Trust estate

The Woodland Trust estate has 652 sites that contain ancient woodland, covering 13,079ha (1.6% of UK ancient woodland). Of this, 189 Woodland Trust sites contain PAWS, covering 3,608ha (1.0% of all UK PAWS).

England

The England Keepers of Time policy has an objective to 'restore or gradually restore the majority of plantations on ancient woodland sites to native woodland by 2030'. Whilst it is positive to have specific mention of PAWS restoration in policy, this ignores the fact that restoration can take decades.

There is also funding available for PAWS restoration in England, however, the amount of PAWS restoration that is being funded via government grants has declined over time with only one hectare reported in 2022-23 (Figure 1). The below figure may be deceptive as it includes PAWS 'worked' (which isn't defined) as well as 'restored'. This likely includes any PAWS that has a management plan or felling licence and isn't necessarily under true restoration.

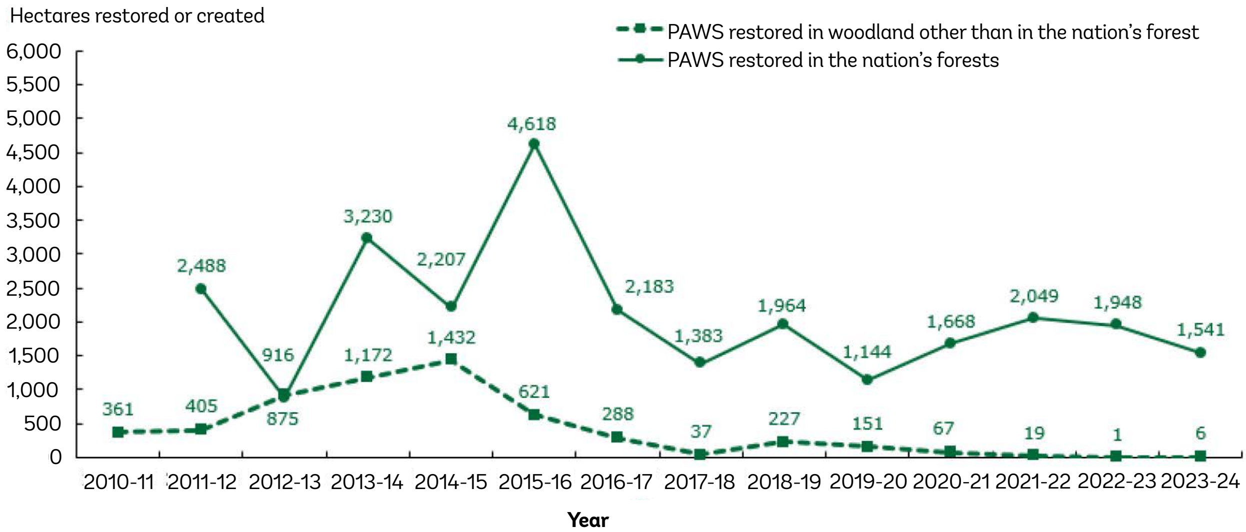


Figure 1: Hectares of PAWS restored or created in England. Source: Forestry Statistics. From: https://www.gov.uk/government/statistics/forestry-commission-key-performance-indicators-report-for-2023-24

However, whilst progress in England is poor, there is at least data being collected on restoration and targets set. Scotland, Wales and Northern Ireland do not publish data on the level of PAWS being supported into restoration or have clear policy targets.

Scotland

The Scottish Government's policy on control of woodland removal states that: *there is a strong presumption against removing ancient semi-natural woodland or plantations on ancient woodland sites, amongst other types of woodland.* Forestry and Land Scotland (FLS) has a PAWS policy set in 2016:

There is over 28,000ha of PAWS in Scotland's national forests. The aim of the PAWS policy is to "protect and enhance ancient woodland remnants and embed them into a native woodland network."

FLS has committed to restoring at least 85% of PAWS to native woodland where restoration is defined as: at least 90% native species with up to 10% non-native naturally regenerating species. The remaining 15% of PAWS areas will undergo enhancing ancient woodland remnants and native woodland features.

Wales

In Wales, priority woodland habitats are defined on the basis of semi-naturalness not on ancientness and therefore mostly exclude PAWS.

In 2012 the condition of the entire c. 19,500ha of ancient woodland on the PFE in Wales was assessed using a combination of field-based sampling and desk-based analyses of both threats and ecological potential. Of this, 34% was considered secure, 36% threatened and 30% critical (pers. comm. Natural Resources Wales, 2020). Approximately 3,250ha of PAWS is larch, the majority of which is likely to be felled in the future due to infection with the

pathogen *Phytophthora ramorum* (see Tree Health section this report). Natural Resources Wales intends to carry out a full repeat ancient woodland condition assessment in 2025.

Northern Ireland

Approximately 50% of the PAWS in Northern Ireland is on the Public Forest Estate, managed by Northern Ireland Forest Service. All ancient woodland sites (including PAWS and ASNW) were surveyed in 2013-14. 709ha were considered 'secure'; not under any widespread threat from impacts such as invasive plant species or shade from non-native tree canopy. A work programme was drawn up for the 302ha of ancient woodland in a threatened or critical condition. The areas identified as threatened or critical in 2013 were assessed again in 2019. The area classified as threatened had decreased by 99ha, and the area classified as secure had increased from 709ha to 809ha. Whilst 194ha remained in a threatened condition, only 9ha was considered to be critical.

Figure 2: The boundary between conifer PAWS on the left and ancient semi-natural woodland on the right photographed at Woodland Trust's Clanger and Pickett Woods before restoration

Despite 40 years of public policy, a considerable amount of the UK's ancient woodland remains as PAWS, and much of this is still likely to be in a critical or threatened condition (Reid et al., 2021). It is also the worst possible moment for a lack of policy targets (as in Scotland, Wales, NI), grant support (Wales, NI) and political prioritisation across the UK as the level of PAWS now at commercial maturity for felling and their future will be decided.

Rhododendron

It is not just plantations that threaten our ancient woodlands. Introduced non-native species such as rhododendron (*Rhododendron ponticum*) has taken advantage of the denuded state of these woodlands and become extremely invasive. Rhododendron presents a particularly formidable challenge, the effect on the wooded ecosystem is profound with the casting of year-round shade, their leaf and leaf litter creating an environment where the remnants of the ancient woodland ecosystem struggle to survive. This continuous shading and disruption halt the regeneration of native trees and plants, and the threats grow progressively more severe if left unaddressed. In addition, Rhodendron acts as a host to *Phytophthora ramorum* (see tree health section of this report), posing a dual threat to woodlands as it also attacks beech, oak and larch trees.

Accurate distribution data for rhododendron does not currently exist at a UK scale, making landscape scale removal of rhododendron difficult. This landscape-scale approach is vital to ensure that once rhododendron is removed, it is not replaced by plants spreading from neighbouring land. Effective landscape-scale eradication of rhododendron removal needs sustained long-term coordination across multiple landholdings. Without co-ordination, control tends towards being piecemeal and opportunistic rather than strategic. Any rhododendron missed in a landscape will continue to be invasive and will likely re-establish into areas where previous eradication attempts have occurred.

The Woodland Trust is continuing to investigate landscape-scale remote sensing techniques to be able to map extent and location of rhododendron and other invasive plants. The long-term aim is to scale this up nationally so that an accurate picture of the scale of the problem can be obtained. This will enable a long-term strategy to be devised to eradicate invasive species from the wider landscape and for this to be fully costed over the decades that this is likely to require.

Also necessary, will be regulation and legislation to stop the widescale sale of *Rhododendron ponticum*, and, where it is retained in ornamental gardens and collections, the law to prevent the spread into neighbouring properties is properly enforced. This will of course also require a considerable public messaging campaign to change hearts and minds. It all starts with knowing the extent of the problem and knowing as accurately as possible where it is.

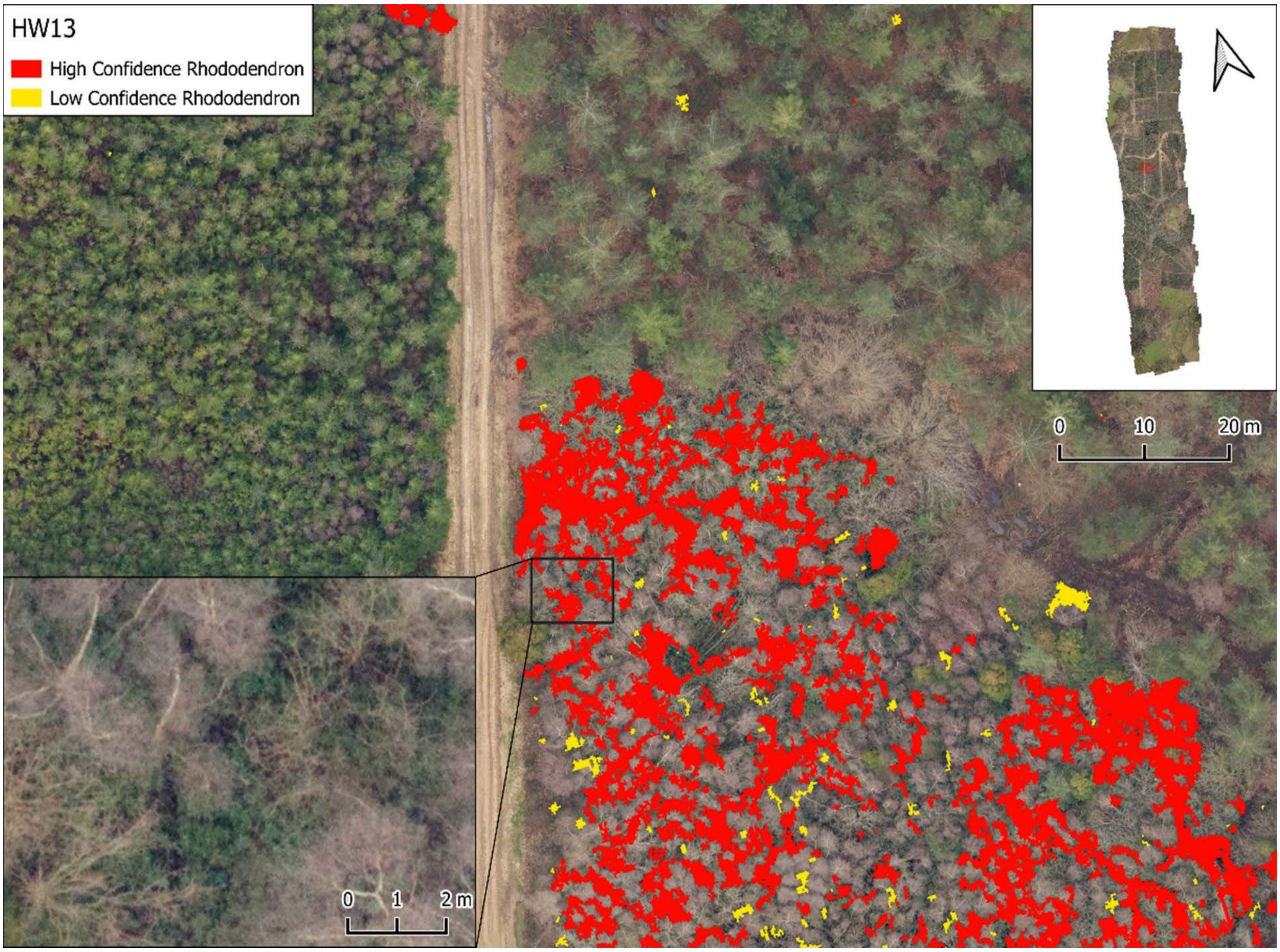
An innovative new rhododendron mapping tool

This project funded by the Forestry Commission's Forest Innovation Fund makes use of cutting-edge advances in remote sensing techniques and explores how these can be used to provide a comprehensive, all-encompassing snapshot of rhododendron in key landscape areas which could be repeated periodically. These maps, in collaboration with others, can be used to strategically plan long-term rhododendron control to make the best use of resources and time.

A coarse overview of 'potential rhododendron' coverage can be provided by satellite data but fine detail hyperspectral data is required to discriminate actual rhododendron coverage. Hyperspectral data is obtained from systems that far exceed what our eyes and standard cameras can see, in terms of the subtlety of information and extent that can be seen. Airborne hyperspectral data was collected (by 2Excel Aviation Ltd) from various sites in the South and the South East of England, during the winters of 2023 and 2024. The data sought to account for geographic, seasonal and environmental variation to develop a general 'rhododendron mapping tool', rather than having a series of site-specific solutions (as some previous studies have provided).

The mapping tool correctly detects the presence of rhododendron (and cherry laurel), e.g. 80%+ accuracy for some areas), although specific problems such as shadow, and the discrimination of a few specific conifer species (that may be confused with rhododendron) required additional analysis stages to separate them from rhododendron. The results are a prediction of areas where there is high confidence that rhododendron is present and others where the presence of rhododendron is less confident, but likely (see figure 3).

In the future, the project team hopes to be able to extend (and test) the versatility of the new rhododendron mapping tool to other areas in Great Britain and use some of the new hyperspectral satellite capability being launched at the end of 2024 to improve the 'potential rhododendron' mapping capability as well.



Ancient woodland soils in restoration

During the restoration process it is important to keep in mind not only the restoration of trees and above-ground biomass but also consider the soils.

It is well-known that the biodiversity below ground far exceeds that above (Wardle, 2002, Bardgett, 2014). Typically, above ground in woodland there will be two or three tree species that dominate, along with a few other less frequent species. Below ground there may be hundreds of different mycorrhizal species, with the roots of an individual tree being in relationship with tens of species (Brundrett, 2009; Tedersoo, 2013).

Typically within a woodland there will be a few mycorrhizal species that are ubiquitous and found throughout. Within this matrix there will be occasions where there will be just a single individual species of a mycorrhizal fungi growing there year after year, and this is the case for the vast majority of species. Rare species may not occur in every woodland even if conditions are similar, and so these are particularly prone to extinction (Dahlberg, 2022).

Much of the uniqueness and richness of our ancient woodlands is in the soils, and therefore protection and restoration of these soils is vital to preserve the ancient woodlands as a whole.

Although ancient woodlands have greater protection under forestry policy, this is mostly around requiring keeping it covered in trees - there is little to no reference to the importance of protecting the soils.

This oversight is understandable, as science is only beginning to uncover the critical roles fungi play in our ecosystems, but also their role in plant survival and adaption at the level of the individual. A greater understanding of the below ground in ancient woodlands and how to sensitively restore with this in mind is needed.

The expanding science and study of the forest soil microbiome is exposing how little we know or understand, and this coupled with the distribution profile of species, means it is likely that many species have already been made extinct by past land use changes, the expansion of unsustainable agriculture and heavy machinery used in forestry. When it comes to managing woodland, we must adopt a precautionary approach to protect the woodland soils, and for ancient woodland soils in particular this should be uppermost in all decisions.

Conclusion

Many of our ancient woodlands have been significantly harmed by plantations of non-native species and/or invasive plants like rhododendron, and many are suffering from multiple other threats, including damage to soils during harvesting.

These disruptions continue to threaten the integrity of ancient woodland ecosystems and now the urgency of the situation, both from plantations and invasive non-native species, cannot be overstated: without real targeted and decisive action, we face the prospect of continuing to lose irreplaceable biodiversity. We are witnessing within our lifetimes the loss and degradation of what is left of much of our precious ancient woodland habitats and species have gone, or are going, extinct locally and nationally.

This necessitates urgent restoration efforts. True restoration involves enhancing ecological integrity and resilience, not merely reverting to past states. Particularly on ancient woodland soils we must adopt regenerative restoration practices over the long term. We must aim to sustain these vital habitats and their biodiversity and improve their condition to be the best they can be. Given the important role these woodlands play in providing habitats for species, storing carbon long term and providing rich biodiversity for people's wellbeing, it is more important than ever to protect and restore them in the changing world.

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Agroforestry

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Abstract

Agroforestry – farming with trees – is an essential tool to help deliver UK biodiversity and net zero objectives on a landscape scale, whilst improving economic resilience and food security into the future. In this non-exhaustive literature review, the beneficial impacts of agroforestry are explored. In addition to the intrinsic value of trees on farm to biodiversity, agroforestry is demonstrated to enhance a range of provisioning, supporting, regulatory and cultural ecosystem services. Agroforestry can offer a 'win-win' in many scenarios, enhancing biodiversity and allowing farming to mitigate and adapt to the effects of climate change, whilst maintaining or enhancing productivity and income. Contextual downsides can also exist, like increased weed burdens, reduced yields of individual crops and increased labour requirements. Widespread monocultural planting of a single species across farmland could also be viewed as an intensive land use and have negative biodiversity outcomes (e.g. on ground nesting birds). However, agroforestry outcomes are site-specific and difficult to predict at a farm scale. Major disincentives remain to the wide uptake of agroforestry by farmers, particularly a lack of financial support and technical knowledge. Greater clarity and integration of evidence in government policy is also needed.

Highlights

- Agroforestry can provide a wide range of ecosystem services, provisioning resources such as food and timber, and supporting and regulating environmental processes.
- In many contexts, agroforestry can deliver 'win-win' scenarios for environmental health as well as farm income and productivity.
- More financial help is needed to make agroforestry an affordable option for farmers as moving to an agroforestry system can require significant outlay and reduce income initially.
- Farmers are often motivated to adopt agroforestry by its benefits for biodiversity and better environmental health, as well as improved livestock welfare.
- A lack of technical knowledge and demonstrations of agroforestry are a significant barrier to uptake.
- Agroforestry can boost rural biodiversity - including birds, invertebrates and plants.
- Government policy needs more clarity and needs to better integrate ecosystem services and

Introduction

Agroforestry can be defined as 'the practice of deliberately integrating woody vegetation (trees or shrubs) with crop and/or animal systems to benefit from the resulting ecological and economic interactions' (Burgess et al., 2019).

Agroforestry is an essential tool to help deliver UK biodiversity and net zero objectives on a landscape scale, whilst improving economic resilience and food security into the future. In this non-exhaustive review, the extent and major benefits of UK agroforestry are discussed, including a broad summary of supported ecosystem services and more detailed examination of the role agroforestry can play for carbon sequestration, biodiversity and livestock welfare. The impact on farm productivity and income as well as barriers and motivations affecting the uptake of agroforestry, are also examined.

Methods

A non-exhaustive literature review was conducted to provide updated information on the extent of UK agroforestry and its ecosystem service benefits. Searches primarily focused on UK studies but also included those from other temperate regions or from a global perspective where necessary. No date restrictions were used, but more recent research (post 2010) was prioritised. Searches for primary evidence were conducted using Google Scholar and included primary research as well as grey literature and websites of key resources or datasets. Research from Woodland Trust-supported projects was also highlighted. In combination with standard Boolean operators, keyword search terms included amongst others: 'temperate' 'agroforestry' 'productivity' 'income' 'biodiversity' 'ecosystem services' 'livestock' 'welfare' 'carbon' 'climate change' 'UK' and 'barriers'.

Results

Definition and extent

Temperate agroforestry can be broadly grouped into two overarching categories depending on if farmland trees are placed within or between fields (Table 1). The components of agroforestry also vary depending on existing land use for forestry or agriculture (Table 1). For trees within fields, a further distinction can be made between silvopastoral and silvoarable systems, with some overlap between the two (agrosilvopastoral) (Table 1). Trees between fields comprise a wide variety of farming systems, including hedgerow and shelterbelt networks (Table 1). Thus, it can be argued that much of the current and historical UK land area exists as an agroforestry landscape. There are few quantitative estimates of the extent or uptake rate of agroforestry within the UK; den Herder et al. 2017 estimate that 2.2% of the UK land area (or 3.3% of total agriculturally utilised area) can be categorised as formal agroforestry, with the vast majority of this in the form of silvopasture (e.g. wood pasture) (~547600 ha). Arable (defined as crops intercropped/or beneath trees) and high value tree agroforestry systems (e.g. fruit orchards) occupy much smaller areas (2000ha and 14,000ha respectively). Quantifying the area of agroforestry depends on the definition and parameters used; for example, den Herder et al., 2017 exclude woody linear features such as hedgerows, which support a range of ecosystem services in farmland, with approximately 477,000km of managed hedgerows in the UK (Carey et al., 2009).

Table 1: Types of agroforestry in the UK and their common components (adapted after Burgess et al., 2019) (*coppiced species include willow, poplar, alder and hazel).

Tree location	Agroforestry system	Land use	
		Forest land	Agricultural land
Trees within fields	Silvopastoral	Forest grazing	Wood pasture Orchard grazing Individual, clumps or lines of trees
	Silvoarable	Forest farming/ gardening	Alley cropping Alley coppice* Orchard intercropping Individual trees
	Agrosilvopastoral	Mixtures of the above	
Trees between fields	Hedgerows, shelterbelts and riparian buffer strips	Forest strips	Wooded hedges shelterbelts and hedgerow networks hedgerow coppice* riparian buffer strips

Ecosystem services

A substantial body of evidence indicates that agroforestry interventions can support a broad range of regulatory or supporting ecosystem services in addition to any marketable outputs provided by trees (Table 2); these include crop pollination (Varah et al., 2013; 2020; Centeno-Alvarado et al., 2023), improvements to soil fertility and health (Dollinger and Jose, 2018), soil erosion control (Torralba et al., 2016; Weninger et al., 2021), crop pest and disease management (Pumaríno et al., 2015; Beule et al. 2019; Huss et al., 2022), flood control (Carroll et al., 2004; Monger et al., 2022) and improvements in water quality (Pavlidis and Tsihrintzis, 2018). In-situ observations also suggest that on-farm woodland and shelterbelt plantings can help to contain or recapture emissions of ammonia (NH_3) and other air pollutants from livestock farming, reducing the potential harm to sensitive habitats in the wider environment (Bealey et al., 2016; Tang et al., 2022) (Figure 1). However, the priority should be to reduce emissions at source. Agroforestry is a key pillar of farm diversification and can provide cultural services such as improved landscape aesthetics, education opportunities and tourism (Table 2) (Phelan & Sharpley, 2011; de Jalón et al., 2018; Herdon et al., 2018; McRae et al., 2024; Pompa, unpublished data). Interviews with farmers also indicate a positive association with improved mental health and a sense of wellbeing (e.g. McRae et al., 2024). There remains a long-standing research bias towards the monetary and biophysical services of agroforestry, with a relative paucity of published studies investigating the socio-cultural benefits (Fagerholm et al., 2016; de Jalón et al., 2017).

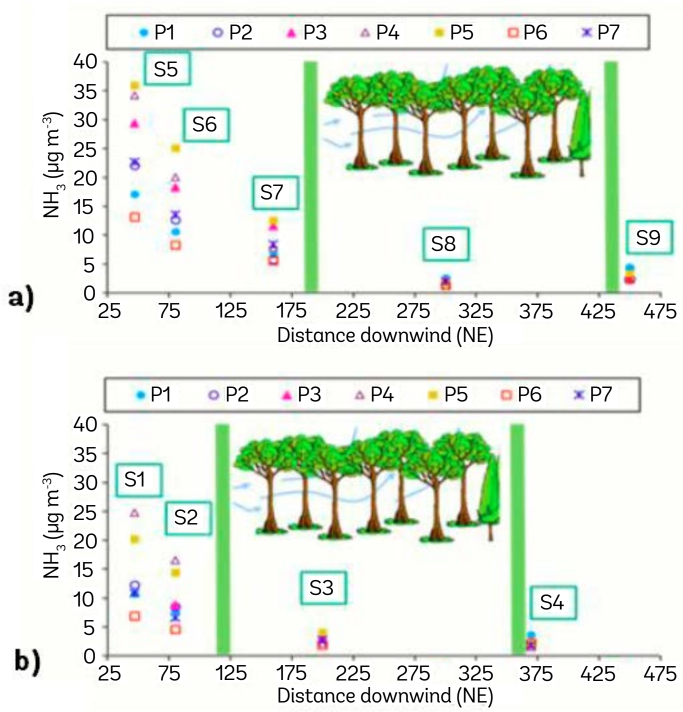

Figure 1: Reduction of dairy farm ammonia (NH_3) concentrations by replanted broadleaf woodland, determined at different sampling sites (s1-9) and time points (p1-7) across 'open' (a) and 'wooded' (b) transects (after Tang et al. 2024)

Table 2: Ecosystem services that can be provided by temperate agroforestry (Varah et al., 2013; 2020; McRae, 2024; Burgess et al., 2019; Staton et al., 2021; 2022, Torralba et al., 2016; Carroll et al., 2004; Dollinger and Jose, 2018; de Jalón et al., 2018; Herdon et al., 2018; Woodland Trust, 2015; Bealey et al., 2016; Lehmann et al., 2020; Monger et al., 2022; Moreno et al., 2017; Beule et al., 2020; 2021; Tosh and Westaway, 2021; Huss et al., 2022; Damianidis et al., 2021)

Regulating	Pollination Soil erosion control Runoff control Storm and flood management Water and air quality Water resource management Shading and microclimate control Wildfire control Biocontrol Carbon sequestration Pest and disease resilience
Provisioning	Food and forage (incl. premium meats) Wood and fibre Fuel Genetic resources
Supporting	Soil formation and retention Nutrient cycling N-fixation Primary production Biodiversity
Cultural	Aesthetics Education and networking Hunting Agrotourism Recreation Sense of place Improved mental health and wellbeing

Carbon sequestration

Agriculture is a major source of greenhouse gas (GHG), amounting to around 11% of UK emissions (DEFRA, 2024). Expanding agroforestry is one essential measure in the path to reach net zero from the UK agriculture and land use, land use change and forestry sectors (LULUCF), as part of a suite of measures including reduced consumption of animal products and improved control of wastes (CCC, 2020). As well as the carbon (C) fixed in timber and root and branch biomass, trees can significantly enhance soil organic carbon (SOC) for depleted and disturbed arable soils (Burgess and Graves, 2022). Detailed modelling, commissioned by the Woodland Trust and performed by Cranfield University (Burgess and Graves, 2022) predicts agroforestry interventions have the potential to sequester 7.0-8.3 t CO_2e ha^{-1} yr^{-1} on cropland, and 6.1-15.9 t CO_2e ha^{-1} yr^{-1} in grassland respectively (Table 3). The highest C sequestration was found for a 40-year rotation 400-tree ha-1 larch silvopasture system (Table 3). Expanding these systems by 10,000ha (<1% of the total area of UK arable or grassland area) would equate to 88-200 kt CO_2e yr^{-1}, with faster rates of planting leading to earlier peaks of C sequestration (Burgess and Graves, 2022). At the most extensive scale, increasing the area of long-rotation silvoarable farming by 10% compared to 2022 levels, whilst establishing new hedgerows or shelterbelts on 11% of UK cropland, would minimise impact on crop yields and enable arable systems to reach net zero by 2050 (Burgess and Graves, 2022; Woodland Trust, 2022). Assuming no other emission cuts in farm operations, establishing silvopasture on 30% of UK grassland area, with a 14% reduction in livestock production in line with meat consumption trends, would also allow pastoral farming to reach net zero by 2051 (Burgess and Graves, 2022). Outcomes of tree planting vary widely by soil type and past use; for example, tree planting on grassland may lower SOC stocks due to loss of perennial grasses (although above-ground biomass can lead to a significant increase in overall C (Upson et al., 2016)). Planting on peat soils also has significant potential to disturb and liberate existing C stocks and should be avoided in most circumstances (Friggens et al., 2020; Woodland Trust, 2023). C sequestration rates also vary widely by species, with fast-growing conifer species such as Scots pine (*Pinus sylvestris*) and non-native larch (*Larix decidua*) generally considered to have a larger potential to fix and sequester C than broadleaf species such as sycamore (*Acer pseudoplatanus*) across shorter time frames. The C sequestration potential of hedges increases significantly if allowed to grow in width and height (Axe et al., 2017). Monocultural planting has significant environmental implications, and agroforestry should reflect multiple objectives as well as GHG mitigation; including biodiversity and profitability (CCC, 2020; Burgess and Graves, 2022). The variability associated with agroforestry may make it better suited towards individual farm GHG budgeting and net zero targets, rather than tradable carbon credit schemes such as the woodland carbon code (Burgess and Graves, 2022).

Table 3: Predicted mean greenhouse gas emissions (negative values) or carbon sequestration (positive values) of modelled shelterbelt, hedgerow, silvoarable (wide alley, poplar 150 stems ha^{-1}) and silvopasture (larch, 400 stems ha-1) systems on cropland or grassland (after Burgess and Graves, 2022)

2022 Baseline (t CO_2e ha^{-1} yr^{-1})	Intervention	rotation (yrs)	Sequestration (t C ha^{-1} yr^{-1})		Greenhouse gas sequestration (t CO_2e ha^{-1} yr^{-1})			
			Soil	Tree	crop & Livestock	Soil	Tree	Total
Cropland	Shelterbelt (6m)	40	0.50	1.72	0.00	1.83	6.31	8.14
-1.87	Hedgerow (2m)	40	0.50	1.66	0.00	1.83	6.09	7.92
	Silvoarable (150 stems ha-1)	19	0.46	1.65	-0.77	1.69	6.05	6.97
	Silvoarable (150 stems ha-1)	30	0.29	2.10	-0.49	1.06	7.70	8.28
Grassland	Shelterbelt (6m)	40	0.00	1.72	0.00	0.00	6.31	6.31
-3.94	Hedgerow (2m)	40	0.00	1.66	0.00	0.00	6.09	6.09
	Silvopasture	24	0.00	4.08	-2.64	0.00	14.97	12.33
	Silvopasture	40	0.00	4.90	-2.05	0.00	17.97	15.92

Biodiversity

Trees on farms increase the range and provision of habitats and niches, microclimates, insect food or nutrient quality and abundance, shelter and egg-laying sites, increase landscape-scale connectivity and buffer habitats (Hewitt, 2022). Through these drivers, agroforestry can significantly enhance the abundance and richness of rural biodiversity and contribute towards the ecological restoration of farmland (Hewitt, 2022). A wide variety of species are known to benefit from agroforestry interventions, including birds, spiders, earthworms, flies, shield bugs, beetles, small mammals and soil microbiota (Hewitt et al., 2022) (Figure 2). This includes threatened bird species such as the song thrush (*Turdus philomelos*), yellowhammer (*Emberiza Citrinella*) (Sage et al., 2006) and barn owls (*Tyto alba*) (Woodland Trust, 2015), charismatic mammals including the hedgehog (*Erinaceus europaeus)* (Yarnell and Pettett, 2020), red squirrel *(Sciurus vulgaris)* (Woodland Trust, 2015) and dormouse (*Muscardinus avellanarius)* (Goodwin et al., 2018) and pollinating insects (Varah et al., 2013; 2020). Specialist species may benefit more in the short term (Staton et al., 2022). Agroforestry produces transitional vegetation communities situated between deciduous forest and farmland. The heterogenous nature of features such as coppice mosaics and understorey vegetation strips can support a greater species richness than agricultural land or high forest (e.g. Müllerová et al., 2015; Kirby et al., 2017; Staton et al., 2021). However, as with many mature woodland habitats, dense canopies or a lack of appropriate management may reduce plant species richness at agroforestry sites over time (Kirby et al., 2017). Alley cropping may also favour perennial creeping species such as creeping thistle (*Cirsium arvense*) due to a lower rate of disturbance compared to conventional systems (e.g. Staton et al., 2021). Agroforestry introduces (or enhances) a tree or understorey-associated microbiome to depleted agricultural settings, increasing fungal or bacterial biomass, which may affect change in soil enzymic diversity and activity (e.g. Beule et al., 2020; Beule & Karlovsky, 2021). There are also potential downsides to agroforestry in some contexts. For example, a large expansion of tree cover may displace ground nesting or farmland specialists such as yellow wagtail (*Motacilla flava*) and lapwing (*Vanellus vanellus*) (Sage et al., 2006). Tree understories or vegetation strips may also be colonised by common arable weed species such as sterile brome (*Bromus sterilis*), blackgrass (*Aloprecurus myosuroides*), ryegrass (*Lolium* spp.) and annual meadow grass (*Poa annua*) (Burgess, 1999). However, an increased presence of arable weed species in tree rows does not necessarily translate into increased recruitment into adjacent crop alleys, or in reduced productivity of the agroforestry system (Boinot et al., 2019; Staton et al., 2022). Agroforestry may shift pest management priorities; for example, fields with alley cropping may experience decreased presence of specialised root flies and an increased presence of generalist slugs compared to conventional arable fields (Staton et al., 2021). Overall, the outcomes of agroforestry on biodiversity are site-specific and difficult to predict, reflecting a range of biotic or abiotic factors including past land use and ongoing management (Hewitt, 2022). The positive biodiversity effects of agroforestry are likely to be greater across a catchment scale, but individual sites can benefit greatly (Torralba et al., 2016; Hewitt, 2022). Meta-analysis also suggests that the biodiversity of arable farms, which often represent the most intensively managed sites in a landscape, are more likely to see benefits to biodiversity than pasture or grassland sites (e.g. Torralba et al., 2016; Mupepele et al., 2021). Aside from the effect of new tree plantings, the high diversity often hosted by veteran or ancient trees in wood pasture habitat across the UK and Europe is well established, and such individuals represent a valuable genetic, ecological, and cultural resource (Moreno et al., 2017).


TOM STATON

Figure 2: A maturing alley-cropping system. Agroforestry systems can enhance biodiversity compared to monocultures and support a range of species including insects, birds, plants and soil microbes

Livestock welfare

A key benefit of agroforestry for UK farmers is in the higher welfare and productivity afforded to livestock, though this is an under-researched topic in temperate regions (Jordan et al., 2020) and most evidence from the UK has been anecdotal in nature or based on grey literature (e.g. Woodland Trust, 2015; Smith et al., 2016; Kendall et al., 2019; Landworkers Alliance, 2021; Robinson, personal communication).

The value of many tree species as browse – including ash, lime and willow - is increasingly recognised, and can provide a supplement of metabolizable energy, fibre, protein, vitamins or essential elements (Emile et al., 2016; Kendall et al., 2021) (Figure 3). Silvopasture can service specific nutritional needs; for example, the use of willow fodder to meet high cobalt and vitamin B_{12} requirements in finishing lamb (Walker et al., 2022). Tannins, and other secondary metabolites provided by tree or plants, modulate immune responses in ruminants and may reduce parasitic burden and the need for chemical treatments (Lira et al., 2008; Rodríguez-Hernández et al., 2023). Introducing tree fodder has the potential to reduce GHG emissions from enteric fermentation, urine or wastes (Ramírez-Restrepo et al., 2010; Stoate et al., 2024). Agroforestry in the form of windbreaks, hedgerows or open silvopasture is also a key tool for managing microclimates in livestock and crop production systems, affording both protective shelter in cold weather or high winds, and shading during summer or heatwaves (e.g. Bird et al., 1998; Tamang et al., 2010; Martins et al., 2021; Atkin-Willoughby et al., 2022; Amorim et al., 2023). Provision of trees increases the thermal comfort of livestock and reduces the incidence of hypothermia and heat stress; this can directly translate into productivity and financial benefits such as increased liveweight gain (Burgess et al., 2019), reduced mortality (Jordan et al., 2020), increased fertility and milk yield in dairy cows (Jordan et al., 2020; Martins et al., 2021), reduced wintering feed or housing costs (Smith et al., 2016 and references within). Trees enhance animal stimulation, decrease stress and encourage natural feeding, self-medicating, grooming and roaming behaviours (Woodland Trust, 2015; Burgess et al., 2019). Trees can also play an important role in managing disease outbreaks by encouraging less livestock clumping and acting as barriers in shelterbelts to separate animals (Burgess et al., 2019).

JAMES ROBINSON

Figure 3: Natural browsing by cattle

Farm finance, productivity and barriers

The multiple advantages of integrating trees on farms has considerable potential to boost productivity and long-term income (Burgess et al., 2019; Lehmann et al., 2020; Pent, 2020; Staton et al., 2022; Forestry Commission, 2023). However, as with other aspects of agroforestry, published evidence and models giving financial costs, risks and benefits for agroforestry remain limited. An assessment of a range of agroforestry systems from across Europe, including the UK, found overall productivity in the form of land equivalent ratios (LER) was boosted 36-100% compared to monoculture systems (Lehmann et al. 2020). Temperate silvopasture, producing a range of products including timber and livestock, can have significant higher productivity when assessed against individual production systems (Lehmann et al., 2020; Pent, 2020; Amorim et al., 2022). Enhanced productivity also translates into arable settings. For example, Staton et al., 2022 assessed productivity and projected farm income for fruit tree alley cropping sites across Southern and East England and compared them to conventional arable systems. Although there was an 11% reduction in cereal yield, caused partially by an increased weed burden, overall productivity and estimated gross mixed income (GMI) for agroforestry systems was substantially greater than in conventional arable systems across the long term (20 years), due to an additional mixed income source (Staton et al. 2022). Furthermore, apple yields for the agroforestry sites per area of tree row were comparable to conventional orchards despite a lower density of trees and may benefit from enhanced pollination rates (Staton et al., 2022). Flowering understories in alley cropping systems may also directly translate to increased farm income through reduced pest burdens and mowing costs, and in countryside stewardship grants (Staton et al., 2021). However, in the assessment by Staton et al., 2022, the time lag for the GMI of agroforestry system to exceed standard arable systems is estimated to range from seven to 14 years (Figure 4), presenting an initial negative cashflow without additional support while the agroforestry system matures. Options analysis suggests a lack of upfront support, as well as potential penalties for GHG emissions and low value of carbon credit, also limits the financial viability of silvopasture (Abdul-Salam et al., 2022). Conversely, break-even points for poultry farms may be met in as little as six months, with rapid improvements in animal welfare and egg quality from tree planting (Woodland Trust, 2015). Break-even points for agroforestry systems are lower if other ecosystem services, such as avoided nutrient runoff and soil erosion, are regularly factored into economic analysis as well as marketable outputs (de Jalón et al., 2017; Kay et al., 2019; Giannitsopoulos et al., 2020). Analysis of perceived disincentives by farmers highlights the central importance of financial constraints, particularly a lack of support for establishment and capital investment (Tosh and Westaway, 2021). A lack of conceptual and technical understanding, uncertainty about agroforestry policy, increased labour requirements during establishment and market uncertainty around novel goods are also barriers to farmers (Tosh and Westaway, 2021; McRae, 2024). Conversely, key motivations for farmers to plant trees include improving biodiversity and environmental health (Pompa, R. unpublished data; Westaway et al., 2024) and other on-farm benefits such as livestock welfare, protection of soils and agrotourism (McRae, 2024). A review of 100 years of UK Government tree planting policy in the agricultural landscapes suggests that, for successful implementation, schemes must not be overly complex or burdensome, have a degree of flexibility, be aligned with farmer values and have a system of support and guidance (Westaway et al., 2023). There is considerable potential to improve existing policy and better integrate frameworks on land use, sustainable development and agroecology (Venn and Burbi, 2023).

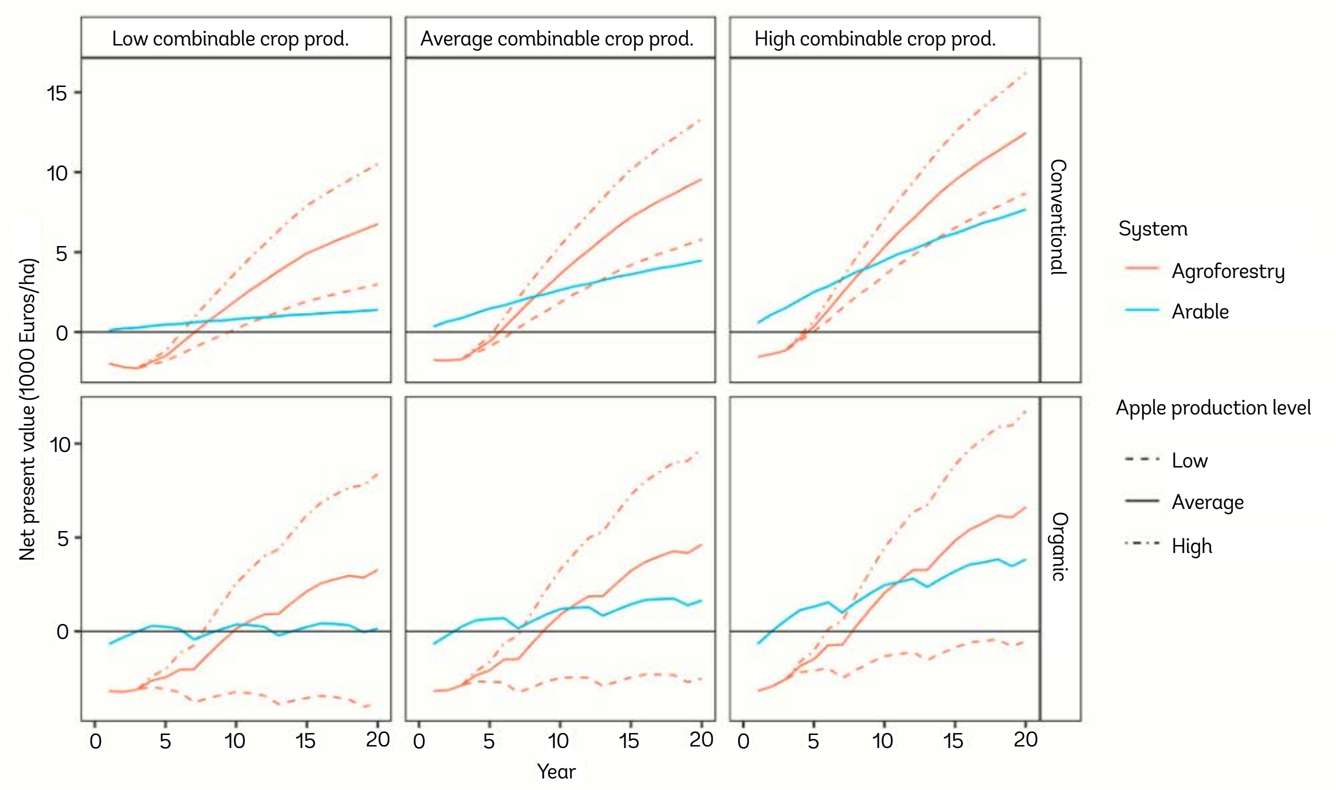

Figure 4: Modelled cumulative gross mixed income (€/ha) and break-even points for alley-cropping agroforestry systems (red line) vs arable (solid blue line) under conventional and organic regimes and variable productivity levels.

The Woodland Trust has been engaged in shaping the new Environmental Land Management (ELM) scheme since it was first announced, sitting on Defra's ELM stakeholder group and feeding into the design of the broader scheme, as well as specific tree/wood options. Through the ELM test and trial programme, the Woodland Trust was a partner in an earlier test looking at barriers to farmers taking up agroforestry and is currently leading a test looking at different approaches to provide advice to farmers interested in the agroforestry options. Throughout, the Woodland Trust has emphasised the importance of trees in delivering multiple simultaneous benefits. As well as engaging on tree-related options, the Trust has been engaged on other ELM options aimed at reducing agricultural emissions, for example the sustainable farming incentive (SFI) nutrient management standard. In July 2024, the Trust welcomed the launch of new SFI agroforestry options but is still keen to see what will be available to farmers and land managers through the Countryside Stewardship higher tier options, which are yet to be published. The Trust continues to engage with Defra on ELM and emphasises the need for actions to be more ambitious to deliver on environmental targets, including around air quality.

Discussion: agroforestry in a changing world

Within the last 10 years, UK temperatures and rainfall have repeatedly exceeded records (Met Office, 2024). Predictions for the UK's future climate suggest a continuing increase in warmer, wetter winters and hotter, drier summers, along with an increase in the frequency and intensity of extreme weather events (Met Office 2022). There is growing recognition from scientific, government and farming voices that agroforestry can play a key role in helping to improve farming sustainability and food security in response to these challenges (Landworkers Alliance, 2021; Mbow et al., 2019; Committee on Climate Change, 2020; Farmers Weekly, 2021; Robinson, personal communication). In addition to the climate change mitigation potential of agroforestry by C capture and storage, trees on farms can be a cost-effective tool to help reduce farmland flooding, lower crop and soil temperature and heat stress in animals, manage water resources during drought and protect soils from erosion and desertification (Carroll et al., 2004; Torralba et al., 2016; Weninger et al., 2021; Jacobs et al., 2022; Monger et al., 2022; Amorim et al., 2023). Agroforestry could also play a positive role in helping to protect crops, livestock and woodland habitats from an expanding range of pests and diseases, facilitated by changing environmental conditions (Bett et al., 2017; Skendžić et al., 2021; Forest Research, 2024). From an economic perspective, the higher productivity and mixed income associated with agroforestry ventures can help to offset future global instability in food prices and provide resilience to rural communities (McRae, 2024). Simultaneously, the ability of agroforestry to restore or enhance natural processes and biodiversity makes a substantial contribution towards ecological restoration and in slowing and reversing the decline of woodland and farmland species (Hewitt, 2022).

Evidence suggests that agroforestry can offer 'win-win' outcomes in many scenarios, helping farmers to restore rural biodiversity and ecological processes, and mitigate and adapt to climate change, whilst maintaining or enhancing farm productivity and income. However contextual downsides exist; without financial support, established markets for novel goods and improved value recognition and pricing of ecosystem services, switching to an agroforestry system can lead to a net loss for farmers compared to conventional systems. Although agroforestry typically increases the overall productivity of farming systems, individual components such as cereal crops are likely to experience reduced yields. Widespread monocultural tree planting across farmland could also be viewed as an intensive land use and have negative biodiversity outcomes if used inappropriately (for example, the effect on ground nesting birds where these are a concern). Overall, despite the well-documented benefits of agroforestry, significant barriers and challenges remain that are limiting its uptake by farmers; funding and technical knowledge gaps are frequently reported as major disincentives. Continued field research into agroforestry is needed, especially clearer demonstrations of economic viability and technical management over time. Greater clarity on the design of the UK Government Environmental Land Management (ELM) schemes and their support for afforestation is also required (Westaway, 2024).

Evidence gaps

- Continued research into the productivity and financial benefits and risks of agroforestry (especially break-even points).
- Technical demonstrations and knowledge sharing.
- Feasibility of alterative or shared management/ ownership contracts to split or share the burden of tree establishment and management.
- Improved valuation/implementation of natural capital/ecosystem services.

- Ecosystem service benefits of hedgerows.
- Impacts of agroforestry systems on biodiversity including understudied taxa such as bats.
- Benefits of agroforestry for farmer wellbeing and mental health.
- Effects of agroforestry on water resource management and flood control.
- Improved mapping of agroforestry across the UK.

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Natural flood management

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Abstract

- It is predicted that climate change will increase the frequency and severity of storm events across the UK, leading to increased risk of flooding.
- Spending on flood risk management is large – over £1 billion spent in England alone in 2021.
- Alongside hard-engineered flood defences, natural flood management (NFM) can provide significant flood regulation alongside other ecosystem service and nature recovery benefits.
- Woodland creation can deliver NFM benefits compared to other land cover types by reducing peak flow rates due to greater interception of rainfall, evapotranspiration, soil infiltration, surface roughness and water storage in soils.
- However, there are still significant evidence gaps around the NFM potential of different woodland types, and under different site and climate conditions.
- Evidence from modelling NFM impacts at a landscape-scale woodland creation and habitat restoration project predict peak flow reductions of 5.3% for a one in 50 year storm event. This is estimated to be worth over £2 million (£12,300±3,300 per hectare of woodland) in flood regulation services over a 100-year period.
- The efficacy of woodland creation to deliver NFM will likely vary with both creation design and local site conditions. It is important that robust site assessment and landscape scale planning is used to prioritise interventions to have the biggest impact.
- Integrating NFM within other nature recovery and ecosystem service objectives has potential to deliver multiple co-benefits and increase our resilience to the impacts of the climate crisis.

Introduction

The latest climate projections (UKCP18) from the Met Office predict that the UK will experience warmer, wetter winters and hotter, drier summers over the coming century, alongside increases in the frequency and intensity of extreme weather events (Met Office, 2018) (see 'extreme weather events' section). This is predicted to increase flood risk across the UK as has been evident in the frequency of flood events over the previous decade (Chatterton et al., 2016; Marsh et al., 2016; Sefton et al., 2021). Increased flood risk may be greatest in the uplands and associated catchments which have experienced the largest increase in rainfall when compared to the lowlands (Burt and Holden, 2010). It's estimated that spending on flood and coastal erosion risk management in England increased from £777 million in 2018 to £1.063 billion in 2021 (Office for National Statistics (ONS), 2023). A breakdown of spending for each of the four nations is limited due to lack of consistent and comparable data.

Alongside increased focus on flood management using hard-engineered flood defences, there has been growing interest in the viability of natural flood management (NFM) to provide flood regulation benefits. NFM covers a wide range of interventions that aim to work with, or enhance, natural processes to reduce the risk of flooding alongside habitat creation, restoration and nature recovery objectives (Lane, 2017; Kay et al., 2019). Examples of NFM include floodplain restoration/reconnection, pond creation, peatland restoration, slowing the flow of water courses using leaky dams, woody debris and stone, and woodland creation. There has been significant investment in delivering and monitoring NFM in England. Between 2017 and 2021, DEFRA funded 60 projects across a £15 million NFM pilot programme. In 2023, Defra and the Environment Agency provided a further £25 million to 40 additional projects as part of a wider target of delivering 260 NFM projects between 2021-2027 (Defra; Environment Agency, 2023). It is estimated that the pilot programme delivered the equivalent of 1.6 million cubic metres of water storage and reduced the risk of flooding to ~15,000 homes (Environment Agency, 2021). The Welsh Government's 'Natural Flood Management Accelerator Programme' was also launched in October 2023 and will see a further £4.6 million invested in NFM schemes throughout Wales covering 23 projects spread across 8 different local authority areas.

Methods

A non-exhaustive literature review was conducted on the role of UK woods and trees delivering natural flood management and flood regulation using Google Scholar to find published peer-reviewed literature as well as grey literature, websites and key datasets. Searches focused on UK studies of native woodlands where possible. Detailed comparisons between native woodlands and non-native plantation forestry were out of scope of this report. Primary search string used was 'forest*' OR 'wood*' OR 'tree* AND 'NFM' OR 'natural flood management' OR 'catchment' OR 'riparian' OR 'peak flow' OR 'flood*'. Reference lists were checked to see if they contained additional relevant sources. In particular, the literature review by cooper et al 2021 and systematic literature review conducted by UK CEH were highlighted as a recent overview of the topic. They should be referred to for additional depth on the topic and available evidence.

Results and discussion

Natural flood management potential of woodland

Woodland creation can deliver NFM benefits by reducing peak flow rates and flood peaks due to greater interception of rainfall, evapotranspiration, soil infiltration, surface roughness and water storage in soils when compared to other land cover types (Stratford et al., 2017; Murphy et al., 2021; Monger et al., 2022). Modelling the impact of woodland creation has also shown that increasing woodland cover can reduce peak stream flows (Buechel, Slater and Dadson, 2024; Monger, D. V Spracklen, et al., 2024), and that the extent and location of new woodlands can impact the size of peak flow rate reductions (Monger, D. V Spracklen, et al., 2024). A natural capital assessment of the flood regulation services that could be delivered by existing UK woodland cover, including trees outside woodlands (ToW), when compared to grassland, estimated the value to be around £12.5 billion (£3,970/ha) over the next 100 years or the equivalent of £420 million/yr (£133 ha/yr) when expressed as an annualised central estimate (Broadmeadow et al., 2023). These estimates are UK averages and the delivery of these natural capital benefits from flood regulation services will vary considerably at different locations with different site conditions and weather patterns.

Cooper et al., 2021, conducted a literature review of the impact of forested land for natural flood management which assessed woodland types across four categories as defined by the UK Environment Agency's Working with Natural Processes – Evidence Directory (Environment Agency, 2018; Cooper et al., 2021).

These categories are:

- Catchment woodland - defined as the total area of all woodland within a catchment, comprising woodland cover of all types.
- Cross-slope woodland - smaller areas of woodland typically placed as belts across hill slopes, broadly following the contours.
- Floodplain woodland - comprising woodland lying within the fluvial floodplain that is subject to a regular or natural flooding regime. This also encompasses riparian woodland.
- Riparian woodland - woodland located directly within the riparian zone, defined as the land immediately adjoining a river channel and influenced by it.

Notable paired-catchment studies such as the Coalburn Catchment in Kielder Forest, and the Plynlimon catchment in mid Wales, have explored the impact of plantation forestry on water yield (the amount of water that runs off the land into water courses) over multiple decades. These studies have been fundamental in informing our understanding of forest hydrology and water use at large, catchment scales and have provided strong evidence that afforestation can decrease water yield (Bosch and Hewlett, 1982) and thus reduce flow peak rates in water courses. However, such paired-catchment studies have also reported variable effects post felling and during the early establishment of forest cover. It is also hard to disentangle the effects of site factors such as soil properties, topography and ground vegetation on adjacent land, which can greatly alter water yield (Birkinshaw, Bathurst and Robinson, 2014).

Research into the effects of cross-slope woodland on peak flow rates has shown that small areas of shelterbelts optimally placed within a catchment can provide significant reductions in peak flow rates (Marshall et al., 2009). Although predicted and measured flow reductions from cross-slope woodland tend to be less than complete catchment woodland cover, research demonstrates that strategically increasing woodland cover in mixed land use settings could greatly contribute to flood regulation services while balancing other outcomes such as agricultural production (Cooper et al., 2021).

Studies exploring the impacts of floodplain and riparian woodland on flow rates are often associated with other NFM interventions such as leaky dams and woody debris (Gurnell et al., 2002; Nisbet et al., 2015), or the impacts on sediment and diffuse pollution from agriculture or infrastructure development (Nisbet et al., 2011; Turunen et al., 2019; Dunn et al., 2022). This can make it hard to disentangle the impacts of woodland cover from other interventions. An additional consideration for floodplain and riparian woodland creation as NFM in the lowlands, is the potential negative impact interventions could have on adjacent land use and communities if measures to slow flow rates increase local flooding or impact existing infrastructure by accumulating woody debris in storm events. Further work is needed to better understand these potential positive and negative impacts and how woodland creation and NFM can be most effectively incorporated into floodplain and riparian zones. This may be particularly important in highly modified agricultural lowland landscapes due to the associate benefits on water quality and mitigating the runoff of sediment and pollutants.

Cooper et al., 2021, concluded that while appropriately planned and managed woodland can reduce flood risk and delay flood peaks, there is a lack of available data for most woodland types with the majority currently being at a catchment scale and plantation forestry. A systematic literature review conducted by UK CEH identified 71 papers (17 from the UK) and found a similar lack of available data especially of observational studies that directly measured flow rates. Overall, they suggest there is evidence that increasing tree cover reduced flood peaks but that the picture was less clear if observation and modelled outcomes were analysed separately with several observational studies finding no significant effect of woodland extent on flood peaks, or in some cases increasing flood peaks (Stratford et al., 2017). There is also a general lack of data from native broadleaved woodland when compared to the data assessing the impact of conifer plantations on hydrology and flood regulation. This is especially the case for large catchment scale studies, with the majority of studies looking at broadleaf woodland being based at a micro catchment or plot level.

The existing evidence reviews on the impact of woodlands on reducing flood risk, highlight the need for a greater number of observational field-based studies across different woodland types that collect broader environmental and contextual data on woodland structure and condition (e.g. tree species, tree spacing, age, management interventions, soil type and properties and land use history). This data would improve our understanding of woodland hydrology and allow us to better assess the generalisability of modelling studies that explore NFM delivery through woodland creation.

Case study

Snaizeholme – natural flood management benefits of catchment-scale woodland creation in the UK uplands

Case study adapted from Monger, D. V. Spracklen et al., 2024 submitted

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IMAGE CREDIT: JOHN CRAWFORD

Introduction

In April 2023, the Woodland Trust started work creating one of the largest contiguous new native woodlands in England - Snaizeholme, near Hawes in the Yorkshire Dales. The project will support landscape-scale nature recovery though the creation and restoration of open and wooded habitats. The project aims to deliver a range of ecosystem services such as the protection and restoration of existing soil and peatland carbon stores, additional carbon sequestration through woodland creation, and the provision of flood regulation services through NFM interventions. The Yorkshire Dales has the lowest woodland cover of any national park in England with total tree cover less than 5% and ancient woodlands only making up 1% of that cover. Large native woodlands are particularly scarce in this landscape which has been highly modified through hundreds of years of woodland clearance, widespread sheep and cattle grazing, alongside the drainage of sensitive wetland and blanket bog habitats.

Snaizeholme is a complex project where habitat restoration and nature recovery are as much a part of the plan as woodland creation to create a diverse mosaic of wildlife-rich habitats. Three phases of planting between 2023 – 2025 will establish 291ha of new woodland, comprising 126.61ha glades (25–400 trees/ha), 70.65ha open woodland habitat (400–800 trees/ha) and 94.15ha groves (800-1600 trees/ha). Predominantly on the upper slopes, montane and sub-montane tree and shrub species will be re-introduced. This is a habitat that has effectively been lost from the Dales and Snaizeholme would be one of the first sites to re-introduce these species, including various dwarf birches, dwarf willows, and bog myrtle. This will provide an important habitat and act as a future donor site for cuttings and seed. 113ha of deep peat will be restored and retained in good condition, 77ha of valley bottom riparian meadows will be managed through low density conservation grazing with native breed cattle, and 81ha of limestone pavement will be managed across a variety of habitats, from sparse scrub, through to conservation grazed grassland.

Snaizeholme sits between 300 and 650m, with a gently sloping valley bottom containing Snaizeholme Beck fed by more than 20 tributary streams (c. 42km of streams). The catchment experiences mild winters and cool summers with mean monthly temperatures ranging from -0.3 to 18.3°C and mean annual precipitation of 1779mm, with monthly rainfall ranging from 88 to 231mm (1981–2010 mean, Shap weather station at 255 m AoD) (Met Office, 2020).

The Woodland Trust site accounts for 92% of the watershed for the upper valley and 42% of the catchment for the whole valley. This provides substantial opportunity to restore woodland and other habitats across a significant proportion of an entire catchment and monitor the impacts on hydrology at a site that is largely independent from adjacent land use, including plantation forestry in the lower catchment. Snaizeholme Beck feeds the River Widdale and then the Ure, part of the SUNO catchment (Swale, Ure Nidd and Ouse) which exit to the North Sea via the Humber Estuary. Immediately to the south, water flows into the Wharfe catchment, and to the south west, the Ribble. Normal flow is around 8-10cm deep on average, although the beck will easily flood to 1-2+ metres.

Since 2022, the Woodland Trust has been collaborating with researchers at the University of Leeds to establish robust monitoring of the hydrology within the Snaizeholme catchment to assess the impact of woodland creation on flood regulation and NFM. This work has included installing in-stream flow monitoring stations across the site, data loggers measuring soil moisture and temperature, and weather stations collecting real-time data to allow us to identify the impact of weather events on flow rates down into Snaizeholme beck. Research at this site is part of a wider long-term project aiming to better understand the impacts of upland woodland creation on ecosystem processes that will be key in the fight against the impacts of a changing climate for people and nature. A first step in this work has been modelling the projected impact of woodland creation scenarios on peak flow rates and flood risk. Here we summarise the key findings from Monger, D. V. Spracklen, et al., 2024 – in prep, to outline this work.

Methods

A rainfall runoff model called 'spatially distributed TOPMODEL' (SD-

TOPMODEL) was used to simulate the impacts of different land cover scenarios in the Snaizeholme catchment on river runoff. This can be used to assess the impact of different interventions or land uses on peak flow rates and therefore potential flood risks for different levels of rainfall events. (full descriptions of methods used in this analysis can be found in Monger, D. V Spracklen, et al., 2024 – in prep).

A range of woodland creation scenarios were simulated to assess the impact of randomly increasing woodland cover in the entire catchment from 13.1% to 58% (including plantation conifer stands in lower catchment) as well as the woodland creation design proposed by the Woodland Trust for this site. The Woodland Trust's creation design was based on a wide range of information and site assessment including the mapping of national vegetation classification (NVC) habitats, archaeology, breeding birds, soils and peat depth. The woodland was designed to enhance existing site features while protecting priority habitats by avoiding areas of deep peat, archaeology and areas frequented by breeding waders. The woodland will be established without the use of tree guards, pesticides or extensive ground preparation, through planting native broadleaf species including alder, silver birch, downy birch, willow, aspen, rowan, hawthorn and blackthorn. The woodland design includes planting at three densities in order to increase structural complexity and the range of woodland habitats/microclimates.

Finally, the potential economic value of flood regulation services delivered at Snaizeholme was assessed using the method of Broadmeadow et al. (2023) which is based on the equivalent capital costs of delivering flood storage through reservoir construction and maintenance.

Results

The modelled flood regulation benefits from the Woodland Trust's creation design reduced peak flow for a one-in-10-year storm event by 5.1% and for a one-in-50-year storm event by 5.3%. This was estimated to provide a flood regulation service worth £2.02±0.54 million or £12,300±3,300 per hectare of woodland (over a 100 year period). This matches or exceeds the market value of carbon sequestration of the woodland based on UK Woodland Carbon Code prices in 2023 and typical carbon storage values for upland sites (Perks et al., 2010). It is also important to note that although the final woodland creation design did not maximise the NFM potential when compared to the greatest modelled reductions in peak flow rates, it balanced multiple objectives and protected existing site features such as avoiding areas of peat and high carbon soils as well as the valley bottom frequented by breeding waders.

The results of this study are likely underestimates of the project's total impact on reducing flood risk as this initial work only modelled the impact of woodland creation areas. Peatland restoration, river restoration and reprofiling, leaky dams and returning woody debris to water courses have the potential to considerably increase the flood regulation impact of the Woodland Trust's work at Snaizeholme. Future research and monitoring will explore these effects alongside ground-truthing the rainfall runoff models using direct in-stream flow monitoring and near real-time weather data on rainfall and storm events.

Conclusion

Woodland creation (catchments, cross-slope, floodplain and riparian) can be an effective natural flood management intervention which can provide significant flood regulation services to downstream communities alongside carbon sequestration, nature recovery and other ecosystem service objectives. Taking a science-led landscape approach to both site design and ongoing monitoring means better outcomes across a suite of objectives (woodland creation, carbon, habitats, access, water management etc.). It also allows us to directly report the impact of our work in a scientifically rigorous and transparent way. This means we can learn from our ongoing work and share our results across the sector.

While there has been some great work delivering woodland creation as NFM and monitoring impacts across the UK, there are still evidence gaps around the role of woodlands to deliver flood regulation benefits. This is true for natural flood management in general. The sector must continue to take an evidence-based approach and be willing to adapt the delivery of our conservation projects in line with emerging evidence and best practice guidance. It is also important to recognise that the use of natural flood management approaches will likely need to be supplemented by other engineered flood reduction measures due to the severity and unpredictability of climate responses. Integrating both approaches will likely provide the biggest flood regulation benefits for downstream communities whilst also delivering a whole host of co-benefits arising from habitat creation and nature recovery.

Evidence gaps

- There is a lack of data on the NFM potential of different types of native broadleaf woodland at large scales.
- Impacts of woodland structure and ecological condition on the provision of NFM benefits are poorly understood. Research in this area could greatly inform practitioner land management, and how to design and place woodlands within catchment to maximise the flood regulation impacts.
- The impact of the largest storm events and the capacity of woodland ecosystems including the soil, to continue to provide flood regulation services before becoming saturated.
- The impact of combinations of different NFM interventions: not much known about how peak flow reductions from woodland creation, peatland restoration, in river interventions (wood debris dams) etc. combine within a catchment (i.e., what is the maximum reductions you could achieve through a combination of interventions).
- NFM under future climate: how does changes in climate (more intense storms, longer droughts, hotter temperatures) alter the role of vegetation and soil in NFM and water retention?
- What impact will increased atmospheric CO2 concentrations have on plant responses such as leaf area, rates of photosynthesis and evapotranspiration?
- The impact of woodland on low flows has received less research attention

and not much is known. There is uncertainty both from both a water management and habitat condition perspective. Woodlands likely increase evapotranspiration and so could reduce annual water yield, and it is sometimes assumed they will also reduce low flows. However, this is not clear and likely to depend on a wide rate of factors including surrounding land use, soil properties/infiltration, impact of shading and surface humidity etc.

- How to maximise NFM in productive forests? Some existing forestry practices such as planning open ground along streams (this is meant to reduce acidification and reduce sediment issues from logging operations) could reduce NFM benefits. Is it better to prioritise non-productive broadleaf (reduced issues with acidification or sediment loss) along streams and water courses (rather than open ground)?

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The scope of this report

This 'State of' report presents important facts and trends focusing predominantly on our native woods and trees. Specific trends and benefits associated with more commercial forestry activities (often non-native plantations) are outside the scope of this report, because they are reported elsewhere. Naturally, these two strands of the UK's treescape are intertwined - there are many links and similarities including drivers of loss and damage, and benefits such as access and pollution reduction. So, this report is relevant across all UK woods and trees.

How we produced this report

The data in this report draws on multiple sources, including official statistics, published and unpublished reports, academic research, outputs from citizen science projects and trends data from regularly updated datasets held by government(s) and non-governmental organisations. We are hugely grateful to them all for sharing their data and time to enable us to present these significant results here.

We have found huge variability in the data available. There are often no equivalent datasets across all four countries of the UK – some cover single countries (e.g. Native Woodland Survey of Scotland), Great Britain only (e.g. National Forest Inventory), and others are UK wide (e.g. wildlife indicator trends). Many governmental organisations and other data providers are devolved across the UK (e.g. the statutory nature conservation bodies) and have adopted slightly different standards and thresholds or delivery methods. Variable baseline dates and coverage can be a challenge, as well as data recording methods changing over time as technology and objectives evolve. Several datasets are incomplete and only record a proportion of the resource (e.g. the Ancient Tree Inventory), yet others were developed several decades ago and have never been comprehensively updated (e.g. the Ancient Woodland Inventory for Scotland). New issues have emerged where data recording is in its infancy (e.g. the impacts of extreme weather on native woodlands), we have relied on case studies to demonstrate the importance of a wider issue and hopefully inspire future recording efforts. This is the second iteration of this report and over time we hope to build knowledge and fill gaps.
We hope that over the coming years gaps in our knowledge can be filled so that we can present an ever more complete understanding of the state of the UK's woods and trees.

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